U.S. DEPARTMENT OF COMMERCE
National Technical Information Service

PB-271 923

Silvicultural Chemicals and Protection of Water Quality

Oregon State Univ, Corvallis School of Forestry

Prepared for

Environmental Protection Agency, Seattle, Wash. Region X

Jun 77

BIBLIOGRAPHIC DATA SHEET	1. Report No. EPA - 910/9-77-036	2.	3. Recipient's Accession No. PB-271 923

4. Title and Subtitle

SILVICULTURAL CHEMICALS AND PROTECTION OF WATER QUALITY

5. Report Date
JUNE 1977

6.

7. Author(s)
OREGON STATE UNIVERSITY

8. Performing Organization Rept. No. N/A

9. Performing Organization Name and Address

OREGON STATE UNIVERSITY
SCHOOL OF FORESTRY
CORVALLIS, OREGON 97331

10. Project/Task/Work Unit No.

11. Contract/Grant No.

12. Sponsoring Organization Name and Address

U.S. ENVIRONMENTAL PROTECTION AGENCY
WATER DIVISION
1200 SIXTH AVENUE
SEATTLE, WA 98101

13. Type of Report & Period Covered
FINAL

14. 910/00

15. Supplementary Notes THIS REPORT WAS PREPARED UNDER EPA CONTRACT #68-01-3553 BY OREGON STATE UNIVERSITY, CORVALLIS, OREGON.

16. Abstracts THIS REPORT IS A COMPREHENSIVE REVIEW OF MANAGEMENT PRACTICES INVOLVING SILVICULTURAL CHEMICALS AND EVALUATES THESE IN RELATION TO BOTH WATER QUALITY AND SILVICULTURAL OBJECTIVES. IT PRESENTS AN ARRAY OF PROCEDURES AND BEST PRACTICES THAT PERMIT REACHING MANAGEMENT GOALS WITH NEGLIGIBLE IMPACT ON WATER QUALITY. THE REPORT DISCUSSES THE GENERAL SCOPE OF CHEMICAL USAGE IN THE PRACTICE OF SILVICULTURE, PRESENTS CRITERIA FOR LIMITING CONCENTRATIONS OF CHEMICALS IN WATER, COVERS BEHAVIOR OF CHEMICALS USED IN SILVICULTURE OPERATIONS AND PRESENTS POLLUTION CONTROL GUIDELINES.

17. Key Words and Document Analysis. 17a. Descriptors

SILVICULTURAL CHEMICALS	FERTILIZER
HERBICIDES	WATER MONITORING
PESTICIDES	CHEMICAL APPLICATION
WATER QUALITY PROTECTION	CHEMICAL CONCENTRATION MAXIMA
TOXICOLOGY	SILVICULTURAL PRACTICES

17b. Identifiers/Open-Ended Terms

METHODS, PROCEDURES, PRACTICES FOR REDUCING WATER QUALITY DEGRADATION FROM THE APPLICATION OF SILVICULTURAL CHEMICALS.
WATER QUALITY MONITORING IN SILVICULTURAL CHEMICAL APPLICATION.
EFFECTS OF AERIAL APPLICATION OF CHEMICALS AND ALTERNATIVE SILVICULTURAL PRACTICES ON WATER QUALITY.
TOXICITY DATA FOR SILVICULTURAL CHEMICALS.

17c. COSATI Field/Group

18. Availability Statement RELEASE UNLIMITED	19. Security Class (This Report) UNCLASSIFIED	21. No. of Pages 235
	20. Security Class (This Page) UNCLASSIFIED	22. Price PC A11/MF A01

FORM NTIS-35 (REV. 10-73) ENDORSED BY ANSI AND UNESCO. THIS FORM MAY BE REPRODUCED USCOMM-DC 8265-P74

U.S. GOVERNMENT PRINTING OFFICE 1977-798-041/155 REGION 10

EPA 910/9-77-036
JUNE 1977

PB 271 923

SILVICULTURAL CHEMICALS AND PROTECTION OF WATER QUALITY

U.S. ENVIRONMENTAL PROTECTION AGENCY
REGION X
1200 SIXTH AVENUE
SEATTLE, WASHINGTON 98101

REPRODUCED BY
NATIONAL TECHNICAL
INFORMATION SERVICE
U. S. DEPARTMENT OF COMMERCE
SPRINGFIELD, VA. 22161

EPA 910/9-77-036

JUNE 1977

SILVICULTURAL CHEMICALS AND PROTECTION OF WATER QUALITY

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This document is available to the public through the National Technical Information Service, Springfield, Virginia 22161.

The Environmental Protection Agency, Region X, has reviewed this report and approved it for publication. Mention of trade names or commercial products does not constitute an endorsement or recommendation for use.

ACKNOWLEDGEMENTS

The authors of this report received data, advice and physical assistance from numerous individuals and agencies, without which the document would have been substantially less complete. In particular, Logan Norris, U.S. Forest Service, Pacific Northwest Forest and Range Experiment Station, George Brown and Robert Beschta, Oregon State University, provided much assistance in reviewing the approach and early drafts of this study. Members of the National Forest Products Association Forest Chemicals Action Committee, chaired by William Lawrence, reviewed the approach and early drafts. Samuel Krammes, U.S. Forest Service, Washington, D.C. provided consultation and large numbers of references used as resource material. George Dissmeyer, Dan Bacon, and Bill Stevenson, U.S. Forest Service, Atlanta, contributed time for consultation and furnished considerable published and unpublished data.

Numerous persons from various offices of the Environmental Protection Agency contributed in various ways. From the Washington office, Tom Burkhalter and Bernard Smale provided consultation, office space and secretarial assistance. Sam Fluker and James Crooks, in Atlanta, furnished data and reports from the Southeast Region. Bill Clothier, Region X, assisted by Duane Tucker and Elbert Moore, coordinated the project throughout and provided assistance, editorial guidance and general support throughout the project.

Special acknowledgement is given to the many reviewers whose painstaking efforts contributed substantially to accuracy and completeness of this document.

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WATER QUALITY GOALS AND SILVICULTURAL CHEMICALS

Purpose

SILVICULTURAL CHEMICALS AND PROTECTION OF WATER QUALITY completes a three-part documentation on state-of-the-art technology to minimize or prevent nonpoint source pollution resulting from activities included in the Environmental Protection Agency's Silviculture Project. The Silviculture Project was organized under the provisions of Sec. 208, 1972 Amendments to the Federal Water Pollution Control Act (FWPCA). The preceding two documents are:

1. Logging Roads and Protection of Water Quality, U.S. Environmental Protection Agency, Region X, Water Div. 1200 6th Ave., Seattle, WA 98101. EPA 910/9-75-007. March 1975.
2. Forest Harvest, Residue Treatment, Reforestation and Protection of Water Quality. EPA 910/9/76-020. April 1976.

Provisions of the FWPCA amendments require that measures be taken to control all types of man-caused water pollution. The procedures for the implementation of this requirement are outlined in Guidelines for Areawide Waste Treatment Management Planning (U.S. E.P.A., 1975). The stated objective of the 208 Program, overall is to "restore and maintain the chemical, physical, and biological integrity of the Nation's waters." To achieve this objective, "it is the national goal that, wherever attainable, an interim goal of water quality which provides for the protection and propagation of fish, shellfish, and wildlife, and for associated recreation in and on the water, be achieved by July 1, 1983."

The format for this report was based on the above Guidelines (U.S. E.P.A., 1975), which identified two focal points as steps in the elimination of non-point water pollution. These are: 1) Identification of "Best Management Practices" for new and existing pollution problems, and 2) Development of a non-point source management program to prevent pollution and promote use of the "Best Management Practices" earlier identified. This document provides a comprehensive review of management practices involving silvicultural chemicals and evaluates these in relation to both water quality and silvicultural objectives. It then develops an array of procedures among the best practices that permit the reaching of management goals with negligible impact on water quality.

Scope of Chemicals in Forestry

Chemicals are used in forests for the protection and maintenance of a resource that takes many years to mature. This production cycle may last through several generations of humans, and many administrations. Applications of chemicals are prescribed for benefits that often accrue to future generations without immediate financial rewards for the landowner. Under the circumstances, the indiscriminate use of chemicals in forestry is extremely rare. When there is a use of chemicals that results in temporary adverse effects on water quality it is generally done with the understanding that there is some risk. Approval of such practices is usually given only when a) damage to the overall forest resource will be reduced by treatment, and b) when no alternative is available for use that is known to be adequate in effectiveness.

Corresponding to the EPA's mandate to control water pollution, the public forest land managing agencies also have mandates to manage the public forest lands for the sustained yield of forest products, wildlife, forage, water and recreational opportunity in perpetuity. Much industrial and private land is managed toward the same goals. The Forest Service has developed management guidelines (Pierovich *et al.*, 1975) in which Best Management Practices are prescribed in a wide variety of management situations. Among these practices, chemicals figure prominently as tools that may be used so as to be consistent with all resource values. The environmental context in which these tools are used deserves some attention.

There are about five hundred million acres of forest, about one fourth of the contiguous states, considered as the forest resource base of the United States. These are exclusive of the areas set aside for other purposes, or too low in productivity to produce lumber and fiber economically. About one-fourth is in federal ownership, and somewhat less is in industrial forestry holdings. Of this resource base less than two million acres, or 0.4 percent, is likely to receive any chemical treatment in a given year.

The commercial forest lands are located in watersheds that supply much of the water used for municipal and irrigation sources. The water is usually of high quality, but natural events cause very large variations in quality from standpoints of both users and aquatic life. The naturally occurring biota in streams have evolved in a history of trauma, and their survival has depended on enormous resiliency and adaptability.

Maintenance of the forest resource has a significant positive effect on water quality. Forest resource management implies a very long-term horizon, and resource management activities affecting water quality have a very brief effect in relation to the effect of the resulting forest during the 20-100 years following treatment. The infrequency of treatment, and the small proportion of land area treated in any given year are important factors to be considered in respect to pollution potential. Many forests that receive treatment to place them under management will not return to derelict status under continuous management, and may never be retreated in the same manner. It is likely, however, that the intensively managed forests of the future will receive chemical treatments of some kind for protection and maintenance, and that adverse effects of such applications can be minimized.

Approach

Meeting the mandates for both environmental protection and commodity production is clearly possible. To this end, guidelines developed in Chapter 6 have assumed that the ultimate goal of resource management is maximum sustainable renewable resource use with minimum total environmental impact.

In the development of these guidelines, the following approach is taken. Chemical use patterns are identified, together with the nature and basis of the problems for which they are prescribed. In order to be sure to include all the major problems, only those practices with significant chance of contaminating water are considered. The toxicological properties of silvicultural chemicals are examined in detail, and water quality criteria are proposed that observe a substantial margin of safety in keeping with the nature of forest watershed biota. Principles of water quality are examined to furnish the basis of use prescriptions so as to avoid contamination levels that exceed the proposed criteria. Finally, a guide has been developed from the above principles that gives the land manager an array of practices permitting the production and protection of forest crops on virtually every acre without causing infringement on water quality as judged by the above criteria. Within these guidelines, the manager may elect among practices in support of a variety of land uses almost to the water's edge.

The Nature of This Report

This document can be used either as a state-of-the-art description of present practices or as a guide for development of management practices. It can also be used as a technical review of the toxicological properties of the important pesticides used in forest management. Chapters 2, 3, and 5, describe the scope of chemical usage and the effects of these practices and their alternatives on water quality and other resource values. Chapter 6 offers a summary of guidelines for use of chemicals in forests so as to maintain water quality without compromising other resource values. Chapter 4 offers a summary review of toxicological properties of major forestry chemicals, supported in the Appendix by a review of specific data on a wide variety of organisms. Water quality criteria are proposed in Chapter 4 so as to provide enforceable standards toward which operators can gear their field operations. The guidelines offered in Chapter 6 are designed so that operators anywhere in the United States will meet the water quality criteria through the use of any of several described management options, at their discretion and as befits the set of problems on the site. A glossary of technical terminology is provided in the Appendix to assist in the interpretation of certain technical details in this document and the supportive literature.

PRIORITIES IN POLLUTION CONTROL, BASED ON HAZARDS AND THEIR CAUSES

The over all goal of water pollution control is to prevent, abate or minimize identifiable adverse affects on all biological and social systems arising from changes in water quality resulting from human activities. Development of control guidelines relating to application of silvicultural chemicals requires an evaluation of the harm resulting from their use and determination of its various causes. An understanding of the causes and effects of pollution can lead to a rational selection of alternatives that prevent or minimize adverse impact, thus facilitating achievement of water quality goals.

Chemicals are used for many purposes in forests. These include herbicides used for vegetation management in reforestation, stand improvement, wildlife habitat and watershed management; insecticides used for protection against insect outbreaks and chemical fertilization to enhance productivity. Most of the chemicals applied on a large scale are applied by aircraft. They may find their way into water either by direct application to water or by migration to streams by surface or subsurface flow.

Hazard occurs when chemicals enter ecosystems in such a way that nontarget organisms are exposed to harmful quantities. Reasonable pollution control strategy depends on identifying the routes of entry and applying materials in ways that minimize their ultimate deposition in waterways.

The scale of human activities in our half-billion acres of commercial forests is large. The legacy of 300 million underproductive acres provides evidence that certain widespread exploitive activities require additional management inputs to avoid loss of productivity. Timber harvest dominates these activities, and compensatory reforestation and other cultural practices are increasing rapidly. The latter are the foci of chemical usage, and the need for such use is often a direct result of earlier harvesting and the growing social importance of timber. The expectation of greater reliance on timber as a renewable resource suggests that increased emphasis on chemicals in the practice of silviculture is inevitable.

Application of chemicals by spraying precludes the absolute freedom from chemical pollution. It is therefore likely that a small quantity of pollution can be anticipated in every major forest region. Intensity of chemical use within regions will vary according to the level of timber production. Chapter 3 outlines in detail the scope of chemical usage in the practice of silviculture, and water quality effects resulting therefrom. Chapters 4 and 5 later consider the consequences of use in detail.

Guidelines are needed for controlling pollutants resulting from use of chemicals for several important reasons: The first is that it is virtually impossible to remove pollutants by other than natural degradation means once they are introduced. The second is that if chemicals are not present in (adversely) significant quantities, no constraints are imposed by pollution that limit the uses to which water can be put. Third, the possibilities of unanticipated side effects from pollution are minimized by preventing pollutants from entering the aquatic systems. Finally, a standard reference for pollution control can be expected to upgrade the general quality of application. It is important, however, that efforts to minimize chemical pollutants do not have the undesirable result of stimulating activities that increase non-chemical pollution. The guidelines suggested in Chapter 6 provide a choice of procedures for managers so that the least disruptive can be selected for any given management system. Thus, relative hazard supercedes absolute toxicity in the ordering of control priorities.

There is a very large body of scientific literature describing behavior of chemicals in forests, toxicity, and effects of economic use on resource management. These data are valuable for developing insight into consequences of regulating practices in various ways, from standpoints both of economics and toxic hazard. But there are no precedents for the implementation of pollution control guidelines in forest management, hence little opportunity to evaluate results. Pollution control guidelines that require substantial changes in operating procedure could be very costly. These costs must be measured both in immediate economic terms and in the long-term social cost, if any, of a change in resource management. These costs must be balanced against known and postulated costs to human and biological resources impacted by such pollution.

Comparisons of tangible costs of hazard and of hazard reduction is one method of ranking priorities in pollution control. Value judgements must also be made regarding intangible assets and unknown effects. These judgements must be interpreted on the basis of current toxicological and monitoring data, of which the latter may dwell within the range of concentrations that produce no observable harmful responses. Clearly, the tangible cost method does not apply exclusively in this instance, yet substantial justification must be given for any proposal to regulate silvicultural activities where there is a chance of adverse economic impact.

At some point, the question ultimately arises whether to tolerate certain levels of known harm to one group of organisms in order to promote or restore the development of others. There are also the unknown potential injuries or gains about which some speculation may have to enter the decision process. In nature, populations of virtually all organisms fluctuate under the influence of others without influence on over-all resource values. Land use patterns in forests have a prolonged effect on stream environment, resulting from establishment, maintenance and harvest of the dominant forest species. The short term effects of the practices used to achieve any given specific objective are often lost in the many years of influence of the resulting forest. Because fluctuations in watershed ecosystems already result from both natural and management-oriented causes, consideration of water quality for a particular few days during the life of a forest deserves special attention. This topic is discussed in detail in Chapters 3 and 4.

The best management strategy is regarded here as the set of tactics which produces the least total adverse effects on water quality. Water quality must be measured in terms of the welfare of aquatic ecosystems and potential water users. Reduction in chemical usage leads to impacts associated with non-chemical alternatives. Therefore pollution control strategies cannot be limited only to those practices dealing specifically with chemicals because non-chemical alternatives may be more harmful. Consideration of water quality criteria must therefore include the over-all balance of chemical and physical pollutants and fluctuations therein.

Management priorities for pollution control strategy in this document have as their targets the following factors, in order of importance:

1) Toxic hazard, with significance measured by degree and duration of harmful effects and their attenuation by recommended procedures or their alternatives (Chapters 3, 4, 5 and 6).

2) Resource tradeoff between aquatic and terrestrial resources, where there is conflict (Chapter 3).
3) Cost of implementation (alternatives listed Chapter 6).

These will be used with the rationale that severe or prolonged toxic hazard or sediment impact are unacceptable. Among the procedures which have low impact, the guidelines will favor those that produce the least possible physical disturbance to the dominant forest or soils, within acceptable economic limitations, while accomplishing the silvicultural objective of maximizing timber or other commodity production. Among the choices that meet water quality and silvicultural objectives, there may be several procedures possible from which operators can be free to choose according to specific objectives, cost and available equipment. These procedures are listed in the guidelines for water pollution control, Chapter 6.

GENERAL SCOPE OF CHEMICAL USAGE IN THE PRACTICE OF SILVICULTURE

Silvicultural chemicals may be categorized in three general groups according to the broad objective of use. In the first group, which includes only fertilizers, the general intent is to increase the over-all productivity of forest ecosystems. The specific intent is to thereby increase productivity of commercial tree species. Similar materials are used as fire retardants; because of their low potential hazard and infrequent use near streams, these materials will not be discussed separately. The second group includes herbicides, and has as its general forestry objective the focusing of forest productivity on selected species. This group ordinarily does not influence innate productivity of ecosystems, but rather channels such productivity into species of some special social value. The last group is devoted to the reduction of losses among commercially important tree species that are already established. These chemicals are the insecticides and rodenticides, whose specific targets are insect and animal pests to protect commercially desirable tree species from being damaged or destroyed. Many other chemicals are used by the forest industries for nurseries, road surfacing and various other purposes beyond the scope of silvicultural operations. This report is limited in scope to those chemicals used in silviculture.

Chemicals are used as a very small but important part of a complex technology that makes up the framework of forest management. Because the use of chemicals in forests is unlike the uses of similar materials in agricultural or industrial applications, it is germane to describe their use in terms of the general system of operation peculiar to silviculture.

Categories of Use and Objectives of Their Use

FERTILIZERS—CHEMICALS USED TO INCREASE PRODUCTIVITY

Fertilizers are receiving limited general use in forests today. There are certain major exceptions, however, which make this group of chemicals the one with the largest total poundage of silvicultural chemicals in use. These exceptions are confined to several major forest industrial corporations and public agencies; the industrial applications comprise most of the tonnage.

The primary objectives of fertilization are to overcome specific deficiencies in essential nutrients for tree growth. These operations are focused in the Pacific Northwest, where nitrogen deficiencies are often encountered, and in the Southeast, where the lack of phosphorus often limits tree growth directly and perhaps indirectly by limiting nitrogen fixation.

Fertilizers are usually applied to large areas by aircraft. Fertilizers, like most other silvicultural chemicals, are applied to a very small proportion of the total commercial forest acreage each year, and applications to a given site occur infrequently. Levels of management on most forest lands have not yet reached the stage at which fertility is severely limiting to economic yield. Thus, silvicultural use of chemical fertilizers is in a very different category from the intensive and general use of the same chemicals on agricultural lands.

HERBICIDES—CHEMICALS USED TO IMPROVE PLANT SPECIES COMPOSITION

This group of chemicals includes a diverse set of herbicidal materials, and a wide range of silvicultural objectives involving control of a wide variety of undesirable herbs, shrubs, woody vines, and trees; the forest "weeds." The nature and extent of problems in forests of the United States will undoubtedly lead to wider usage of herbicides than of any other group of chemicals.

Use of herbicides to enhance forest productivity rests on the general principle that the productivity of a forest ecosystem is normally fixed. Economic and social potentials of the ecosystem are based on the abundance, growth rate and desirability of the dominant species. Where adequate stocking of desirable species is maintained by judicious management, there is little need for herbicides. Target species for herbicides are restricted to those plants that use resources otherwise available to desirable species, without providing tangible benefits. The same principle is used in weed control that is observed in harvest-caused deterioration of species composition: i.e. the removal of dominant species promotes development of those that are subdominant. The forests of the United States have been subjected to removal of desirable species for up to 300 years. Unnaturally large components of undesirable species are widespread. Where weed trees and shrubs are now dominant, the composition of such deteriorated stands will remain relatively poor until selective removal of weeds again favors desirable species. It is to this end that herbicides are used. The variety of herbicide uses ranges across the diversity of composition classes in the commercial forests of the entire nation.

It is important in describing herbicide usage to identify a unique aspect of their use in the forest: herbicides affect only primary producers (green plants) when applied at registered use rates. Despite a lack of direct effects on other organisms, they influence all organisms that depend on any or all of the affected primary producers for food, cover or energy sources. For this reason, herbicides are the most powerful management tools at the forester's disposal. They may be used for controlling not only species composition and stand structure, but also have the potential for reducing damage from wildlife, diseases, and insects. Because of their wide spectrum of effects, use of herbicides may be anticipated anywhere, and for almost any purpose.

Most of the herbicides used in forests are applied by aircraft. Rates of application are within the ranges used in agriculture, and application is usually limited to one or two treatments early in the life of a stand. Most of the herbicides used in forestry are registered also for agricultural uses, and have established residue tolerances in foodstuffs.

Few forest stands require general treatment after desirable species become dominant. Stands placed under management may not require treatment in the present sense after future harvests, because the weed problems of the past will not normally recur. Herbicides are now usually used to correct historical problems of fire, non-reforestation or cull hardwoods, and their use is presently generally limited to one short period in the life of a forest when a forest is placed under management. As intensity of management increases, future uses will undoubtedly occur in other patterns, however.

Application of herbicides in or near streams raises questions related to water quality. Timber growing sites adjacent to streams often are among the most productive, and the application of herbicides near creeks, particularly by aircraft, invariably results in some quantity being deposited in the water. Buffer strips along watercourses are often left unsprayed in order to reduce deposits in streams. This is the principal pollution control technique for aerially applied herbicides. Drift control nozzles and spray adjuvants have also been used to reduce herbicide deposits outside the target zones.

In contrast to the case with fertilizers, there is widespread use of herbicides with ground application methods, including mist blowers, conventional power sprayers and tree injectors. The latter method will account for an increase in usage of herbicide in the decades to come as hardwood forests of eastern United States are placed under managements but quantities applied by these methods are presently small. In terms of future impact on water, injected herbicides are of academic interest in relation to water quality. This topic has received considerable attention in relation to the organic arsenicals, but measurable contamination has not been found (Norris, 1974). Further discussion of movement of herbicides not applied to open water will be developed in Chapter 5.

INSECTICIDES AND RODENTICIDES—CHEMICALS USED ON ANIMAL PESTS

These chemicals are of particular concern in relation to aquatic systems on account of their status as animal toxins. The insecticides, in particular, are targeted mainly on widespread epidemics of defoliating insects affecting large continuous areas of commercially valuable timber. Regional projects often include entire river drainage basins in general aerial applications.

Values of insect-vulnerable timber are often high; where treatment is warranted, there is economic incentive for treating all infested stands. Unsprayed areas risk direct economic loss; they also can lead to residual breeding populations of insects capable of reinfesting sprayed stands.

The materials registered as insecticides for aerial application, with the exception of endosulfan, are short-lived organophosphorus and carbamate insecticides. Endosulfan is used principally on local insect outbreaks in Christmas trees and ornamentals.

Non-residual biological agents are also used for controlling insects. These suspensions of insect disease cultures are quite specific for insects. Their effectiveness on insects and lack of known impact on non-target species are encouraging. Nuclear polyhedrosis virus cultures have been used effectively in several projects with considerable success and demonstrably low impact on non-target terrestrial and aquatic species. This approach is now registered for use on the Douglas-fir tussock moth. Even the most selective of the chemical insecticides, however, are harmful to certain aquatic insects and food chain species at concentrations below 0.1 mg/l (see Appendix for specific data).

Ground applied insecticides and rodenticides are used in such small quantities in forests that their total effects on water quality are not likely to be detectable. Areas tend to be small and isolated by untreated strips from open water. Application rates of insecticides are typically low in relation to agricultural uses.

Quantities of Chemicals Used in the Practice of Silviculture

The quantity of a chemical used is important in two major respects. The rate per acre influences the intensity of direct immediate contamination. The total amount of chemical used in a zone or region and its distribution in time and place determine the potential for long-term and large-scale contamination of large water-courses. The former generally affects the likelihood of acute toxic hazard; the latter determines the potential for chronic damage to aquatic systems and to large scale users of irrigation water.

Chemicals applied to forests from aircraft are often used at dosage rates similar to those used in agricultural operations. Movement into water supplies is much less, per pound applied, however, because of low soil movement. Their infrequency of use, and the intact status of most forest soils and vegetation at the time of application minimizes movement. Fertilizers and herbicides, in particular, are usually also applied in a patchwork fashion some distance from users of potable or irrigation water. A given rate of application therefore tends to have much less impact on water quality than would a comparable dosage applied to farmland. A proportion of forested streams in treated areas normally contains detectable amounts of chemical immediately after application. Over all, however, forest uses of herbicides and fertilizers have low impact per unit of use because the observed concentrations are considerably below those known to affect aquatic organisms adversely (Frederiksen et al., 1973), and applications are diffused in time and place

Insecticides are usually applied in very large regional projects in which every watershed may be treated. Even though the same considerations hold as for herbicides in regard to intact forest floors and vegetation, there is more likelihood that large insecticide operations leading to a low level of contamination in small streams will affect major river systems.

Total amounts of chemicals used in forests are difficult to estimate. Sales are not segregated according to use, but major industrial and public user groups maintain records for a large part of the intensively managed forest areas. Programs, especially of insecticides, vary considerably from year to year. Table 1 lists a summary of Forest Service and industrial uses of pesticides for one year to illustrate scope of forest usage.

Effects of Forest Chemicals on Water Quality

Chemicals presently used in forests enjoy a remarkable safety record in many respects, including maintenance of stream productivity. This generalization and the exceptions to it are fundamental to the development of a set of pollution control guidelines. The basis for the generalization will be considered first, followed by a technical review of pertinent data.

A large body of data has been gathered relating to concentrations of herbicides and fertilizers in water. Relatively few data are available regarding concentrations of organophosphates and carbamates in water; much of the insecticide research was focussed on the now-defunct DDT. In short, no recognized reports of injury to stream life have come to the attention of the writers relative to properly applied herbicides or fertilizers in forestry. Reports of general injury to fish have been confined to applications of insecticides, principally chlorinated hydrocarbons, in large projects. Most were prior to isolation of major streams from application patterns. Fish kills have been restricted generally to localized stretches of water and have been followed by recovery over periods of a few months to 3 years. At least one incident, however, was reported in which substantial fish kill was associated with an organophosphate (Hatfield, 1969). Unfortunately, the expertise needed to evaluate impacts on inconspicuous food chain species has been lacking for most projects. Evidence for low impact has often been restricted to a small number of operations. Observations that have been made are consistent, however, and are not in major conflict with laboratory data.

Table 1. Reported pesticides used for silviculture in the United States, 1972. Compiled from U.S. Forest Service Pesticide Use Reports and a National Forest Products Assoc. Survey representing approximately 32 million acres. Actual amounts are considerably greater. Herbicide use does not fluctuate greatly, but insecticide listing for the one year is not representative.

Herbicides	Acres Treated
2,4-D	278,905*
2,4,5-T	189,517*
Paraquat	31,076
Picloram	18,735*
2,4-D & 2,4,5-T	14,907*
2,4-D & Picloram	13,343*
Amitrol	6,979*
Dinoseb	5,082
MSMA	3,671*
2,4-D & 2,4-DP	3,405*
Stoddards Solvent	3,196*
Atrazine	2,664*
Cacodylic Acid	1,920*
Simazine	1,773*
2,4-D & Dicamba	1,305*
2,4,5-TP (silvex)	1,073*
Dichlorprop	882*
Dicamba	857*
Dalapon	800*
2,4-D & Amitrole	580*
2,4-D & Silvex	400*
Diphenamid	329*
2,4,5-T & Dicamba	276*
2,4-D & Atrazine	252*
Amitrole & Simazine	242*
Amitrole, Bromacil & 2,4-D	57
Amitrole, Atrazine & Simazine	30*
Amitrole & Bromacil	26

Insecticides	Acres Treated
Zectran	650,450[1]
Carbaryl	111,691*
Mirex	6,320*
Malathion	5,186*
Lindane	4,154*
Chlordane	693
Benzene hexachloride	569
Thimet	355
Guthion	347*
Phorate	150
Gardona	115
Dimethoate	90
Di-Syston	73

Rodenticides	Acres Treated
Endrin	44,081*
Chlorophacinone	10,110*
Strychnine	632
Zinc Phosphide	97

Fungicides	Acres Treated
Ziram	9,107*
Thiram	8,427*
Methyl bromide	4,320*

* Registered for use in forests or ornamentals, 1976.

[1] Zectran is no longer being manufactured.

Concern with the possibility for harm to aquatic ecosystems from herbicides and fertilizers prompted inquiry of water quality specialists from several state and federal agencies in the Pacific Northwest, southwest and Washington, D.C. In the absence of a pattern of harmful effects in the literature, these specialists were polled to determine if there were "worst cases" on which to draw some inferences of value in silviculture.

The Nonpoint Sources Section of the Environmental Protection Agency, Washington, D.C., reported no knowledge of any instances of aquatic ecosystem damage directly attributable to presently registered uses of herbicides and fertilizers in the practice of silviculture (Drs. Bernard Smale, Thomas Burkhalter, personal communication). Similarly, the U.S. Forest Service Division of Environment Research, Washington, D.C., reported no instances of damage, although there are many records of detectable concentrations of chemicals (Dr. Samuel Krammes, personal communication). The U.S. Forest Service, Southeastern Region, has made observations relating to water quality after fertilization and herbicide applications. Forest fertilization with phosphate in the South has not measurably influenced water quality (Dr. George Dissmeyer and Mr. Dan Bacon, personal communication). Objections to herbicide applications have been filed in various national forests in the South and Pacific Northwest, but there is no record of injury to persons, wildlife or aquatic systems when these chemicals have been used in accordance with registered use rates (U.S.D.A., 1975a, 1975b, 1975c). These observations alone are inadequate evidence for a conclusion that harmful effects do not occur. They are consistent, however, with the recorded contamination levels and toxicity data discussed in detail in Chapters 4 and 5, and in the Appendix. One recorded incident in Oregon involved the accidental placement at an unknown point of a large quantity of 2,4,5-T upstream from a salmon hatchery. In an unpublished report to the Oregon Fish Commission, Newton and Norris (Oregon State University, 1965) reported sustained levels in hatchery ponds of more than 0.1 mg/l for more than one day. Although there was no obvious mortality pattern , possible effects of the herbicide may have been obscured by heavy silt loads from an earlier storm.

The Southeastern Region of the Environmental Protection Agency similarly had not observed aquatic system damage from silvicultural use of chemicals. Dr. Sam Fluker, Chief of the Pesticides Branch, Southeastern Region, indicated in an interview that the water quality problems related to pesticides in the South were tied to direct application to open water or to bare soil. Soil movement with heavy rains was described as leading to deposit of the chemicals in water as they move with silt. Silt-associated pesticide movement was not thought to be a problem in silviculture. Dissmeyer, (1973) however, reported that major deposits of silt had an adverse impact on water quality when non-chemical methods were used in the South for site preparation. If used in combination with application of persistent pesticides, such methods would deposit pesticides as well as silt in the water.

In the Pacific Northwest, no damage reports were found relating aquatic ecosystem damage to toxic action of herbicides. There apparently have been instances in which oil carrier used for herbicides may have produced slicks which caused localized fish kills in standing water (Dr. James Harper, Oregon Department of Fish and Wildlife, Portland, Oregon, personal communication).

Fertilizers have been studied in relation to the contamination of streams with urea, nitrate, ammonia and nitrite. The review by Moore (1975b) of stream contamination resulting from fertilization observed that a very small proportion of urea applied to a forest watershed finds its way to the streams. At the time of application, some chemical falls directly into the stream, and a pulse of elevated concentration moves downstream. Levels of nitrogen fertilizers or their metabolites in streamwater do not apparently approach maximum safe levels for streamwater as recommended for potable water (U.S.D.I., 1968).

Indices of quality in relation to nitrate depend on the uses to which the water will be put, and the type of drainageway the water must follow. Potable water containing high nitrate concentrations is regarded as hazardous to babies, and is limited to 10 mg/l of nitrogen as nitrate for this reason. Streams containing high nitrate levels are also implicated in eutrophication of ponds and lakes; a similar problem exists where phosphate concentrations are sufficiently great to stimulate development of the nitrogen-fixing blue-green algae. The levels of fertility necessary to cause eutrophication are not fixed, however, and are dependent on the rate of turnover in lakes, temperature and other factors. Variations in native nutrient levels appear to equal or exceed the range associated with fertilizer use in forests (Brown _et al._, 1973; Moore, 1975). Thus, nutrient levels found in forest watersheds are probably of minor significance in eutrophication of rivers having few impoundments or lakes. This topic will be discussed in detail in Chapter 5.

Herbicides have received considerable attention in relation to effects of aerial sprays on water quality. Norris and coworkers (Norris and Moore, 1970; Norris _et al._, 1965, 1966a, 1966b) have systematically monitored forest watersheds treated with phenoxy herbicides, amitrole, picloram and atrazine. This work has been undertaken under conditions of operational use where buffer strips have been both present and absent. In terms of toxic hazard to wildlife, herbicide concentrations seldom occur at a peak level equal to those known to cause harm to fish or other wildlife. Furthermore, the observed peaks of concentration have generally persisted for a matter of minutes or hours, dropping to the detection limits within two days after application. An important exception to this pattern was observed in a rangeland treatment of a marshy area with shallow, slow-moving water (Norris, 1967).

Herbicides are specific in their activity on plants. Concentrations of herbicides in water are generally lower than those known to cause injury to crops through irrigation water (Bruns et al., 1972, 1973, 1974). The one important exception to this is that picloram will cause injury to potatoes at concentrations of 0.001 mg/l or lower (Ogg and Wapensky, 1969), and is thought to be equally injurious to tobacco. Picloram is also among the more mobile herbicides in soil water. There is therefore some risk to these highly sensitive crops any time irrigation water is used from creeks downstream from areas in which brush control has been accomplished with picloram. Other crops are not known to be sensitive at these rates, and levels of picloram found in streams are probably a hazard only in connection with irrigation of potatoes, because tobacco is seldom irrigated.

A discussion of effects of forest herbicides on water quality is not complete without mentioning 2,4,5-T. This compound is probably more widely used for conifer release and site preparation by aerial spray than all other herbicides combined (much of the 2,4-D is applied by ground methods). This herbicide has been in use for this purpose for more than 20 years, without recorded mishaps involving humans or wildlife from operational use. Since 1969, when 2,4,5-T and its contaminant 2,4,7,8 tetrachlorodibenzo-para-dioxin (TCDD) were publicised as teratogenic, this herbicide has been very closely scrutinized for harmful effects, with thus far negative findings even though formal experimentation to this effect has been limited (Day _et al_., 1975). There are very few pesticides whose record has undergone such close investigation without some evidence of untoward environmental damage. Ross[1] has indicated that more is probably known about toxicological properties of 2,4,5-T than is known about any other pesticide. Despite this, 2,4,5-T is on EPA's list of pesticides considered to have rebuttable presumption against reregistration. Silvex, another phenoxy herbicide containing TCDD and which is also used as an aquatic herbicide, has toxicological properties almost identical to those of 2,4,5-T. It is used for aquatic weed control in lakes and ponds at registered rates 20-2000 times as high as the maxima observed after application in forests, with no buffer strips. Reports of harmful effects to fish or wildlife from proper operational use of this herbicide have not been found in this review, despite the confining nature of such bodies of water.

Insecticides and rodenticides are animal toxicants of concern for their direct effects on fauna in streams. Insecticides may cause injury to fish at concentrations usually much lower than are required for damage from herbicides. There has been much evidence of fish kills with aerial application of insecticides, DDT in particular, in forested watersheds (Kerswill, 1967). There is also evidence that such applications need not cause injurious levels of contamination if the material is applied under strict supervision so as to keep it from falling directly into the creeks (Kerswill and Edwards, 1967; Tracy and McGaughy, 1975). Chlorinated hydrocarbons can produce effects through biological magnification. In this process, microscopic organisms collect fat soluble compounds to higher concentrations than are present in the water. As these organisms are used for food by larger organisms, the consumers accumulate an increasing body load until the rate of intake drops or until the rate of degradation equals intake at equilibrium. This mechanism gives rise to considerably elevated concentrations of certain pesticides in predatory fish and birds. Barring mortality, these high pesticide concentrations are reversible. At the worst, impacts of chlorinated hydrocarbons can have an effect on anadromous fish runs extending up to three years (Kerswill, 1967). The worst offenders in this respect, DDT, aldrin, dieldrin and heptachlor are not registered for use in forests. At best, pesticides presumably have no effect whatever when concentrations remain below the threshold of toxicity to the most sensitive organisms.

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[1] Ross, Ralph. Director, Dioxin Implementation Program, EPA, Washington, D.C. Address presented to the Western Society of Weed Science, Portland, Oregon. March, 1976.

It is noteworthy that chlorinated hydrocarbons have been replaced for forest insect control almost entirely by chemicals of the carbamate group, (carbaryl) and of the organophosphate class, (malathion, fenitrothion and trichlorfos). These compounds have very short biologically active lives, and are remarkable for their broad spectrum effects on insects, with low toxicity to mammals and sometimes to fish. Aquatic insects are very sensitive to these materials, as are terrestrial target species. There can be a substantial problem with aquatic insects and crustaceans when insecticides, even of the selective group, are applied to open watercourses. With care in application, however, Kerswill and Edwards (1967) and Shea (Dr. Patrick Shea, U.S. Forest Service, Pacific Southwest Forest and Range Experiment Station, Berkeley, personal communication, 1976) have reported no adverse effect on fish with the use of malathion and phosphamidon.

Ground applications of all classes of forest chemicals have traditionally involved such small amounts of chemical at such great distances from open water that their impacts on streams are of academic interest.

Relation Between Forests and Other Watersheds

The potential for water pollution from silvicultural use of chemicals is very much influenced by the nature of the forest systems from which water flows. Certain features of forest watersheds tend to increase the likelihood of pollution from application of chemicals, and others clearly decrease the risk of pollution. The typical features of forests that are important in this respect are listed in Table 2.

Table 2. Features of forest watersheds that increase or decrease the impact of stream contamination from silvicultural activities compared to agricultural or other non-forest watersheds.

Features That Can Increase Impact	Features That Can Decrease Impact
1. Disturbed soils	1. Infrequent application
2. Steep slopes	2. Continuous ground cover
3. Upstream from most users	3. Minimum surface runoff or sheet erosion
4. Frozen soil during spring runoff	4. High rates of dilution
5. Skeletal soils	5. Low proportion of area treated
	6. High aeration
	7. Active soil matrix

The general lack of mobility of the chemicals used in forests reduces the significance of the physical features of forest watersheds that tend to increase contamination; the difference between movement of a few inches or a few feet has little bearing on water quality in the absence of surface runoff. Details of chemical behavior in the forest environment are discussed in Chapter 5 of this report.

In the Western states, forests are principally restricted to headwaters. Water is essentially free of synthetic pollutants in the absence of chemical applications. Elsewhere in the United States forests are intermingled with farms and population centers. There is therefore more opportunity to pinpoint a source of pollutant in the West than in the East, but a given input in the settled portions of the East is more likely to push an existing pollution level above a hazard threshold.

Chemicals and Multiple Use Resource Values

Commercial forests produce a variety of products and benefits among their many resource values. Wood products, water, wildlife, livestock grazing and personal pleasure are the principal benefits to society.

Chemicals are used in forests principally for the management of commercial timber, on which society has placed a premium value. Nearly all the revenues accruing from the management of forested lands come from the sale of wood products. The relation between timber and other resources is somewhat different on public and private timberlands, but commercial timber growth is the dominant use on most commercial forest lands under deliberate management. Under the circumstances, it is germane to consider the question of whether the use of chemicals for the specific promotion of the timber resource compromises other values.

The resource value most likely to be influenced by the use of chemicals appears to be that associated with recreational use. The specific feature of recreational use that may be jeopardized is the potential aesthetic or psychological impact of damaged flora or fauna. Recreational users of forests are often unaware that the intermediate objective of using the chemicals is, in fact, to favor the growing of certain trees or that the long-term outcome is often to restore and maintain species composition and stand structure to forms more amenable to multiple use than exist at present. It is not widely understood that the species appearing as targets are normally those whose abundance have been inflated by human activity, and whose control will restore or protect a prevalence of aesthetically desirable stands. Adverse aesthetic effect is normally of brief duration, and bears no direct relation to water quality unless managers are moved by public pressure to use non-chemical methods of greater impact. Other resource values, including wildlife and water, are affected by management goals but are not obviously influenced adversely by the use of chemical tools for implementation.

The nature of the practices is basic to a discussion of resource values relative to a group of practices. As summarized earlier in this chapter, fertilizers increase productivity of the whole forest ecosystem, herbicides modify the composition and structure of the plant communities and animal toxins influence composition through management of consumption selectivity and rate. Of these, herbicides have the least potential for causing direct harmful effects to aquatic fauna (see Chapter 5). Yet herbicides initiate the restructuring of entire terrestrial ecosystems from which waters flow. These changes are permanent as long as they are maintained by a management system oriented toward timber production. They are not associated with any particular impacts on water quality, although there are slight short-term increases in water yield attributable to decreased transpiration (Newton and Norris, 1976).

The long-term nature of forest responses to herbicides is of fundamental importance in relation to other resource values. Changes in stand composition and structure influence every terrestrial organism that depends on primary production. These, in turn, must therefore have some effects on the aquatic community in relation to leaf and litterfall, insect populations and other factors. Newton and Norris (1976) have outlined some of the principles involved in evaluating the long-term ecological significance of using herbicides.

The impact of insecticides on streams has resulted both from direct effects on fish, and from indirect effects through ingestion of contaminated insects and other fish. In the case of chlorinated hydrocarbons, which tend to accumulate in fat, there has been biological retention and magnification, especially in lake populations (Muirhead-Thomson, 1971). In freshwater flowing systems, the productive capacity of streams is affected only briefly or not at all by low level contaminations by the transient insecticides (Elson and Kerswill, 1966). There is evidence also that fish, as well as insects, can develop resistance to toxic levels of pesticides through multiplication of resistant individuals. The observed recovery of stream productivity after various kinds of disturbances suggests that there are several mechanisms, including avoidance and resistant life stages, through which the resiliency of the aquatic system prevents the prolonged depression of biomass after pulse-type inputs. Observational data indicate that long-term losses do not occur with non-persistent pesticides (Grant, 1967; Symons and Harding, 1974). A presumed exception would occur where decimation of smolts would reduce returns of anadromous fish in subsequent years. A serious resource loss can occur, if streams produce contaminated fish, or valuable runs of anadromous fish are reduced. In this sense, insecticides, especially chlorinated hydrocarbons, have the maximum potential for causing harm to aquatic resources.

Trade-offs between resource values must necessarily enter into prescriptions for water pollution control. It is beyond the scope of this report to make specific value judgments regarding the relative importance of each "commodity" produced by a forest. When recommendations are made regarding treatment alternatives, however, an attempt will be made to acknowledge the consequences both in terms of aquatic and terrestrial resources.

Resiliency of Forest Ecosystems

Forested watersheds consist of stream systems influenced principally by geology and the microenvironment conditioned by the forest cover. In evaluating priorities for preserving one resource at the expense of another, should such a question arise, it is appropriate to evaluate the duration of response when both forest and aquatic systems are responding to chemical inputs.

Ecosystems are resilient. In the classical sense, succession of some sort follows any disturbance to a forest or stream. The ultimate stage of succession, the climax, prevails when each part of the ecosystem is replacing itself in perpetuity. A very long time is required for climax forests to develop, and it can be regarded as unlikely that extensive climax forests have ever existed for long without disturbance by fire or some other agent. Prevalence of subclimax or earlier seral communities is a natural phenomenon, and human activity has increased the percentage of forest areas in the earlier seral stages. This relationship is similarly true of streams, but the shorter successional period is reflected in the shorter life span of aquatic organisms.

During the recovery of a forest from disturbance, changes are likely to occur in forest streams. Total deforestation tends to increase streamflow for several years, after which transpirational losses return to their former levels. Continuous devegetation of a forest results in mineralization of soil nutrients, for which the retention system has been impaired because of reduced demands by growing plants, especially trees. Such effects presumably can result from repeated application of non-selective herbicides or traditional farming practices. Regardless of the method, loss of the retention system appears to be the key to nutrient loss, and the result is an effect on water quality through nutrient enrichment (Likens et al., 1970), or siltation (Dissmeyer, 1973, 1974). Disturbances of this kind have not been a part of practices developed for sustained yield of forest products. Because the general intent of silvicultural use of chemicals is to enhance forest productivity, primary producers are typically abundant enough to utilize nutrients and water without serious losses of nutrients as the ecosystems respond (Miller, 1974). There is some evidence that immature second-growth forest cover releases less nutrients than very young or mature stands (Leak and Martin, 1975).

Resiliency of forest stands also has an influence on forest stream temperatures. Water quality is a function of temperature as well as chemical composition, and streamside vegetation has a substantial effect on water temperature (Brown and Krygier, 1970). Choice of methods for vegetation control influences streamside cover. Aerial application of selective herbicides leaves some streamside vegetation even when buffer strips are not specified. Riparian species typically have rapid growth rates in early years and shade reestablishes itself quickly over small streams. Both mechanical and chemical control of forest vegetation can be effected with minimal effects on such vegetation. If either has an effect on streamside vegetation, however, the recovery of vegetation after application of herbicides tends to be more rapid than after the more complete suppression which results from scarification. Scarification (and its effects) are easier to monitor, however, and accidental treatment of streamsides is more likely with aerial applications of herbicides than with scarification. Whether chemical brush control immediately adjacent to streams has a measurable effect on water temperature has not been studied, although change in shade has predictable effects. With either method, it would appear that the effect of partial control of streamside brush would have a substantially lower long-term effect than would the permanent removal or reestablishment of coniferous cover. That is, the management goal itself likely exerts an influence greater than that of the practice of implementation.

Effects of insecticides on insects and crustaceans can have subsequent effects on fish under some circumstances. Aquatic insects make up a significant proportion of fish diets at certain times of the year. Eradication or reduction of such species has the potential for a sharp reduction in certain fish foods. This may lower carrying capacity in the absence of alternative food supplies, and persistent insecticides may biomagnify so as to cause injury to fish (Sprague _et_ _al_., 1971). Because of the widespread nature of some insecticide operations, there is some possibility that a pulse contamination of chlorinated hydrocarbons, will be followed by relatively slow re-invasion (Muirhead-Thomson, 1971). There is also minimum opportunity for fish to escape to other stream branches or main channels of lower contamination levels. Thus, moderate contamination with a persistent insecticide, on a widespread basis, can reduce carrying capacity for a season or more, although Tracy and McGaughy (1975) observed no reductions after use of DDT. Severe contamination by many insecticides will cause fish mortality directly, but concentrations of short-residual insecticides high enough to cause heavy insect mortality on a widespread basis are unlikely to harm fish even with narrow or negligible buffer strips when there is close surveillance of operations (Tracy and McGaughy, 1975; Kerswill, 1967).

In summary, insecticides of both organophosphate and carbamate groups tend to have pulse effects on streams, with principal impacts focusing on aquatic insects and crustaceans. The long-term effect of a contamination level that might be associated with a non-buffered application by aircraft, i.e., up to .1 mg/l for one day, would be limited to a brief decrease in fish production, lasting less than one year. Because of the lack of biomagnification, it is unlikely that higher trophic levels or organisms would eventually carry potentially harmful residues of insecticide. Organochlorine insecticides, however, have the potential for appearing in streams over an extended interval after application, leading to a more or less chronic exposure. Although it appears unlikely that the long term productivity of any given stream will be seriously reduced by a single application at forest use rates, harmful concentrations could remain in fish and consumers of fish beyond the year of application.

Herbicides are generally low in toxicity to insects, crustaceans and fish relative to insecticides. There is no evidence that measurable adverse effects to fish occur with broadcast applications of presently registered herbicides, nor that other wildlife using the water is likely to be affected. Because of the lack of initial observable effect on the aquatic ecosystem, the principal resource value potentially jeopardized by herbicides relates to irrigation; the aquatic ecosystem itself is resilient enough to accommodate a substantial contamination without major harm being done.

To summarize the comparative generalized effects of insecticides and herbicides on forest resources, the following statements appear to be consistent:

1. Aerially applied insecticides of short residual life may reduce the impact of defoliating insects on forest trees for 2 years or more, and perhaps several decades, if applied to forests not yet severely defoliated, and if the insect epidemic is not on the verge of collapse. Impact on streams without buffer strips, but where application to open water is avoided, risks a reduction in carrying capacity of streams for part of one year, but appears to have no other adverse effects on the aquatic system; the degree of impact varies among insecticides. Treatment with buffer strips reduces the contamination levels in streams, hence minimizes degree and duration of lost productivity of aquatic ecosystems.

2. Aerially applied organochlorine insecticides have short-term effects on terrestrial insect communities similar to that of the short-lived insecticides. They have a longer effect on both terrestrial and aquatic species, which, however, can be expected to recover. Ordinarily, the effects on primary producers is similar - negligible - to those of carbamates and

organophosphates. Effects on stream biota are more persistent when major contamination occurs. Reduction in stream productivity in such instances may last a year or more, depending on the nature of the stream system and the proportion of watershed treated.

3. Surveillance of application will minimize insecticide deposit in streams, hence toxic hazard, regardless of chemical used.

4. Application of herbicides for conifer release, or to prepare for establishment of conifers, causes a change of forest type that persists until the stand is removed. In the Pacific Northwest, this usually means restoration of the species that was prevalent before development of the plant community being treated. Because the conifers becoming dominant as the result of treatment tend to remain dominant, ecosystem resiliency does not restore the system to its pre-treatment status. Since an increase in conifer production is the objective of treatment, this particular ecosystem response precludes the requirement for additional major disturbances until the next cycle of harvesting. With the exception of apparently minor changes in stream habitat due to increased short-term radiation resulting from hardwood defoliation, aquatic ecosystems do not appear to be influenced either in composition or productivity, regardless of whether buffer strips are used. Long-term shading will increase as conifers become dominant, and changes in litterfall into streams will occur.

Problems and Practices Entailing Aerial Applications of Chemicals, and Alternatives

Forest managers confronted with pest or fertility problems often can choose among several alternatives that will support long-term objectives. Most forested areas have never received the direct application of any silvicultural chemical, and many areas in all likelihood will not be treated in the forseeable future. The following summary will outline the most common problems for which chemicals are generally used. Based on data available in 1976, the various solutions to the problems will be listed with their relative costs per unit of effectiveness and relative impacts on water quality.

NITROGEN DEFICIENCY

There are basically two approaches for improving nitrogen nutrition on forest lands. One of these, widely tested and utilized, is the aerial application of nitrogen fertilizer. Urea is the form of nitrogen generally used because of the relatively low cost of chemical and application. A potential alternative is the use of nitrogen-fixing plants as a cover crop. The latter method has not been utilized in any major industrial applications, but deserves more attention in view of low cost and prolonged increment of nitrogen in large quantities.

As of 1976, prices for urea nitrogen are in the range of $20-30 per hundred pounds elemental nitrogen, including application and overhead. Responses can be expected on N-deficient sites for five or more years after applications of 200 pounds N per acre (220 kg/ha). The decrease in response with time is apparently attributable to ecosystem utilization of the added nitrogen;

the findings of Moore (1975) and others (Klock, 1971; Maleug _et al_., 1972) demonstrate that very little of the nitrogen leaves the site in leaching and streamflow. Thus, addition of nitrogen fertilizer appears to add to ecosystem nutrient capital without having appreciable effects on long-term stream quality.

The use of nitrogen-fixing plants involves a complex technology. In the Pacific Northwest, Newton _et al_., (1968) found that red alder could fix approximately 300 pounds of nitrogen per acre per year (330 kg/ha). In order to capitalize on the added fertility, however, Douglas-fir must be free of overhead competition, and the simultaneous establishment of alder and Douglas-fir is seldom attainable. Some nitrogen can be realized without serious suppression of conifers if the dominant alder is sprayed with 2,4,5-T as a release treatment between the fourth and sixth year after establishment (Newton, M., Oregon State University, unpublished data), or if the fir is planted substantially before alder (Newton _et al_., 1968). Release is less essential if very large western hemlock seedlings are planted. The presence of heavy alder competition during the establishment years of either species causes a long-term loss of growth, however.

There is fragmentary evidence that Scotch broom, a leguminous shrub, is a vigorous nitrogen fixer that can be managed so as to fix nitrogen while not suppressing conifers (Newton, M., unpublished data). Broom is regarded generally as a noxious weed, but has a growth habit that does not threaten established Douglas-fir plantations if established two or more years after planting of conifers. Being intolerant of shade, it has the potential for fixing nitrogen before being suppressed out by conifers, without herbicide release. Should broom overtop a slow-growing plantation, it is very sensitive to dormant applications of 2,4-D ester in oil. Other legumes, black locust and lespedeza, in particular, have been used for nitrogen fixation in the rehabilitation of coal spoils in eastern United States. These legumes also must be planted long enough after the crop species so that they do not suppress the species forming the forest crop.

In any program involving nitrogen fixation, certain requirements must be met for the fixer to do its work effectively. Nitrogen fixation requires an abundance of phosphorus, and adequate quantities of cobalt, zinc, manganese, nickel and iron. Soil acidity must be in suitable range for nitrogen fixing species being used, and adequate infection rates of symbiont must be on hand prior to nodulation. The growth habit of the fixer must be matched to the crop species to prevent suppression, and the sensitivity of the nitrogen fixer to herbicides must be understood before it is introduced into a plantation. Many of the nitrogen fixers have the potential of becoming serious weeds as well as making valuable and energy-saving contributions.

Nitrogen fixation is an energy consuming process. Fixation is not an automatic phenomenon in the presence of nodulated plants. Newton _et al._, (1968) observed that fixation tends to decrease in the presence of adequate levels of nitrogen. Red alder fixes nitrogen to a considerably higher equilibrium level than snowbrush ceanothus, but both appear to fix at some rate in proportion to the degree of nitrogen deficiency.

In summary, nitrogen can be applied either by chemical fertilizers or by biological fixation. Nitrogen fixation does not necessarily minimize the use of chemicals; it does entail different chemicals. Fixation often requires phosphorus, minor elements and herbicides. It also demands an exacting comprehension of ecological programming so as to gain nitrogen without losing the crop species. Under ideal circumstances, biological fixation can provide more nitrogen per dollar than can nitrogen fertilizer. It will undoubtedly be many years before the technology of cover crops for conifers becomes operationally feasible on large acreages. Choice of procedure is not likely to have a major impact on water quality. The higher soil nitrogen levels associated with biological fixation will probably be offset by lack of direct application of chemicals to water.

PHOSPHORUS DEFICIENCY

Until recently, there was no commonly used remedy for phosphorus deficiency other than addition of phosphorus fertilizer or soil drainage in certain wet lands. There is now some evidence that certain mycorrhizal fungi may enhance availability of phosphorus (Marks and Kozlowski, 1973). Liming of acid soils and application of sulfur to alkaline soils can also increase availability of the existing phosphorus supply (Buckman and Brady, 1960).

MOISTURE DEFICIENCY ATTRIBUTABLE TO HERBACEOUS WEEDS

Plantation establishment on droughty sites is often restricted by herbaceous vegetation. Newton (1964) reported that herbaceous cover accounted for 85 percent of the total spring and summer losses of moisture in a clay loam soil in western Oregon. Although evaporative losses are relatively greater on coarse textured soils, drought stress can be reduced to tolerable levels on a wide variety of forest sites by controlling herb cover (Newton, 1970).

Control of herbaceous weeds can be accomplished either by chemicals or mechanical equipment. Grazing has been attempted, but stocking levels compatible with seedling survival do not provide for adequate removal of transpirational surface (Black and Vladimiroff, 1963). Chemicals include those of the s-triazine group and dalapon for grass control, and 2,4-D for broadleaf weeds. Application rates for the triazines range from two to four pounds of active ingredient per acre; dalapon is applied at 3.4 to 8.5 lbs/acre (3.8 to 9.4 kg/ha) of active ingredient as the sodium salt; 2,4-D is applied at up to 4 lbs/acre (4.4 kg/ha), usually as the low-volatile ester. These herbicides are generally not applied to disturbed soil except in Christmas tree plantations. They may be applied either by ground or aerial equipment. Prevention of water pollution may be achieved by reducing application to open water and by eliminating tillage. Precision of application by aircraft can be improved by addition of drift-reducing adjuvants, but any procedure that increases droplet size may reduce the effectiveness of herbicides that act through foliar activity.

The effectiveness of chemical weed control is attested by its wide acceptance in the Pacific Northwest, especially in Christmas tree operations. The cost of chemical weed control in that region ranges in the vicinity of $15-30 per acre, ($37-75/ha) in return for which plantation survival increases from a range of nil to 50 percent to a range of 50 percent to nearly perfect. That is, weed control has made possible the establishment of plantations in one operation on sites that have heretofore been left nonstocked.

Mechanical weed control entails the use of a variety of devices calculated to suppress vegetation long enough to permit seedling establishment prior to recovery of weed cover. Conventional ploughing and disking prior to summer fallow and fall disking is one common method. This procedure will provide reasonable survival of planted seedlings; success is enhanced by the use of residual herbicides to prolong the weed-free period. Operational costs are similar to those of chemical weed control when chemicals are not used, and survival tends to be somewhat lower (Newton, 1970). This procedure is limited to sites that have been previously cleared of stumps and rocks.

Other methods have been used in forest situations. Bulldozing to scarify sod is one method used occasionally in the Pacific Northwest; roller choppers and shearing blades have been used in southeastern United States for a variety of weed situations. Mechanical weed control methods do not bury all seeds deeply, and annual weed communities are reestablished with great rapidity unless surface soil is removed. Thus, mechanical methods are not generally recommended for herbaceous weed control because of the short term of their beneficial effect on the plantation. Surface soil disturbance also increases movement of silt into streams. Costs of scarification in grassy areas range somewhat higher than chemical control; roller choppers are approximately comparable to chemical control in cost, but are not generally used in herbaceous weed problem areas. Cost per unit of survival is least for chemicals by a substantial margin in severely droughty areas, with advantages decreasing as summer moisture becomes less critical.

Hand scalping, the process of removing a circle of weeds from a planting spot during the process of planting, has been used extensively. This practice is of marginal value on severely droughty sites, and is labor intensive, hence costly. The use of spot applications of herbicide toward the same end is more effective and less costly; spot spraying leaves a mulch of dead material to protect soil from evaporative stress.

Water quality can be affected by mechanical herbaceous weed control. The combination of tillage and chemical weed control for several years, as practiced in some northwestern Christmas tree plantations, leaves opportunity for development of erosion pavements, with attendant siltation. When done without cultivation, soil structure remains intact, and erosion is much diminished. An analogy to no-till farming is appropriate.

On flat ground, the use of heavy equipment for forest weed control without concurrent use of chemicals probably has very little effect on water quality as long as soil is only lightly disturbed. Machine operation can compact soil and lead to decreased infiltration (Froelich, 1974). But undisturbed forest soils usually demonstrate excess ability to absorb rainfall, and small, temporary decreases in infiltration capacity are apparently possible without causing surface runoff under Pacific Northwest conditions. The short period of weed control by light scarification leads to rapid stabilization of soil surfaces soon after the onset of rains. Herb cover protects against the formation of an erosion pavement. Siltation can occur if heavy equipment operates on steep slopes, in creeks, or pushes debris into channels. Various states already have prohibitions against these practices. The chief shortcomings of the method where it is legal, are its lack of effectiveness when applied with minimum impact on soil, and its destructiveness on soils and watersheds when applied for maximum clearing.

In summary of herbaceous weed problems, chemical weed control is the most cost-effective method in areas of seasonal drought. When used alone, there is negligible deposit of herbicide in water, and siltation is not increased. When used in combination with tillage, siltation can occur if weed control is maintained for a period of several years. In view of the tendency for most herbicides to adhere strongly to soils of high organic content, (Moore and Norris, 1974) a substantial movement of surface particles in forest soils would likely cause the movement of a certain amount of adsorbed herbicide into the stream. The effect of such pollution is unknown, but appears to be slight. Mechanical methods are of some utility, especially in areas where droughtiness is of marginal intensity. Mechanical methods have a maximum impact on soils, perpetuate the herbaceous weed community and risk the direct deposit of sediment and debris in stream channels. Hand scalping is not a satisfactory substitute for chemical or mechanical weed control except in very favorable site conditions. All methods, applied effectively, lead to development of a forest cover, with its attendant long term implications for high water quality.

SHADE AND LITTER PROBLEMS–BRUSH AND HARDWOOD COMPETITION

Undesirable woody plants constitute the largest group of pests affecting timber values in the United States. Walker (1973) reported that some 300 million acres of forest land (60 percent of the commercial forest resource area) were supporting less than 70 percent stocking in desirable species because of weed problems; over 100 million acres are either poorly stocked or nonstocked. Weed trees and shrubs occupy many of these sites in a semipermanent fashion (Newton, 1973) and restoration of productivity will necessarily entail the removal or control of large areas of brush and weed trees. Specific operations involving woody plant control are stand improvement, release and site preparation.

There are several widely used methods for treating brush and weed tree problems, including chemical, non-chemical approaches, and combinations of chemicals with mechanical procedures and fire. These methods produce different effects, depending on the situations in which they are used, and differ in cost. The selection among alternative practices is dependent on the specific nature of the problem. Water quality has not been an important factor in selection, and there have been few, if any, reports showing negative impacts on aquatic systems associated with chemical or mechanical weed control. In recent years, there has been an increase in the use of fire in forest rehabilitation. Fire has been identified with increases in surface soil movement, in some instances associated with development of a hydrophobic layer at the surface.

Fires used in rehabilitation are planned for high intensity and short duration. High soil surface temperatures for long periods are avoided deliberately. Chemical treatment for pre-burn desiccation is a part of fuel preparation that permits burning when large fuel is still moist and adjacent ownerships are low in flammability. Roberts (1975) has reported that revegetation occurs very rapidly after brushfield fires in the Oregon Coast Range. Her work suggests that light burns in nitrogen-rich sites are unlikely to leave soil surfaces vulnerable to excessive acceleration of soil movement. Herbicides, such as phenoxy herbicides at rates up to four pounds per acre, picloram at one pound per acre, and the toxic product dinoseb at up to five pounds per acre, are essential to allow burning when adjacent areas are low in flammability.

Chemical site preparation entails the use of herbicides alone for preplanting treatment of brush. Phenoxy herbicides, picloram, amitrole and ammonium ethyl carbamoyl phosphonate are used for this purpose. Of these, the latter two are remarkable for their low impact on aquatic systems, but their activity on woody plants is somewhat restricted. Glyphosate, at present unregistered, shows substantial promise for this use.

Felling of weed trees is an approach to control that has received widespread use. Felling alone provides adequate composition control and release where existing stands have a sufficiently large component of desirable stems. Sprouting stumps tend to compromise plantations established at the time of felling, however. Chemicals such as phenoxy herbicides and picloram are used to treat stumps to prevent sprouting. The felling of large cull trees also has a severely injurious effect on residual desirable trees. Eastern hardwoods, in particular, are easily scarred. Disease control considerations in stand improvement operations favor chemical treatment of standing trees because the procedure avoids such damage in felling. Girdling can be substituted for chemical treatment in species with low sprouting potential, at considerable extra expense over that of injection (Beavers, 1957). In the only major study of water pollution associated with tree injection, Norris (1974) reported no increase in streamwater content of arsenic when entire watersheds were treated with organic arsenicals by injection for precommercial thinning. The same pattern can be expected with phenoxys and picloram. Even though very large areas of commercial forest land are in need of stand improvement, a choice between injection and felling or girdling is unlikely to have a measurable effect on water quality.

Aerial application of phenoxy herbicides, 2,4,5-T in particular, is a prevalent method of providing release. Applications are scheduled from early spring through fall in the Pacific Northwest, in summer in the

Southeast, and in early August in New England (Fitzgerald et al., 1973; Gratkowski et al., 1973; Newton and Smith, 1976). Chemical release is usually done by helicopter, and the herbicide may be applied either in water or diesel fuel. An increasing acreage is being treated in the South by air blast equipment.

Felling has been used for forest release operations. Costs for release by aerial application of 2,4,5-T are approximately one third of those of hand felling. Damage to suppressed trees by falling hardwoods is a major problem and individual removal of hardwood crowns from bent-over conifers is a major contributing cost in hand release work.

Water quality has been monitored extensively in the Pacific Northwest to determine amounts of herbicides reaching water supplies during aerial application of herbicides. The use of buffer strips and close surveillance of applicators are the principal methods of pollution control. Patterns of herbicide deposit observed in streams and their behavior are discussed in detail in Chapter 5 of this report.

Treatment of brush in buffer strips by non-chemical means is presently not generally feasible in the steep forested topography characteristic of western United States. The use of mechanical equipment on the occasional gentle topography increases both cost and risk of siltation. Hand felling is possible, but costly and ineffective in reducing sprouting. The stability of untreated brushfield communities is well documented (Newton, 1973; Newton et al., 1968; Gratkowski et al., 1973) and leaving non-stocked brush untreated essentially deletes those areas from the timber resource base. The consequences of various streamside management schemes are important considerations in the choice of practice in management of buffer strips.

DEFOLIATION BY INSECTS

Insect outbreaks occur in patterns that differ substantially from vegetation problems. Epidemics often cover large contiguous areas or a high concentration of foci within an outbreak area.

Aerial application of insecticides is a standard procedure for minimizing insect damage. The prescription of aerial broadcast application of insecticides therefore entails the risk of pollution on major stream systems. Nearly all insecticides currently used for aerial application of forest insects in the United States are of the organophosphate or carbamate groups. Applications are sometimes repeated over extensive areas to control spruce budworm in Maine and eastern Canada, but seldom elsewhere in the United States. In all areas, use of buffer strips is the prevalent method of pollution control.

Alternatives to chemicals include biological insect control agents, predatory insects, attractants and silvicultural control of insect habitat. The sterile male technique has not been adapted operationally to forest insects. Integrated control systems entail combinations of chemical, biological and silvicultural control techniques.

Biological insecticides include _Bacillus thuringiensis_ and nuclear polyhedrosis virus suspensions. These are applied in the same manner as insecticides, with substantial effectiveness on several insect species. Both are in the emerging stages of technology, and a virus has been registered for use on the Douglas-fir tussock moth. Their advantages include high specificity for target insects and lack of residue; their disadvantages include high cost, occasional failures in maintaining culture viability, and amount of lead time needed to have adequate fresh culture on hand for operational treatment of large epidemics. The potential effects of biological agents on water quality have not been investigated at the same level of intensity as chemical pesticides, and consequences to aquatic systems are unknown, but apparently minimal.

The use of predatory insects in the classical biological control sense can be shown to be effective on a local basis. Logistical problems and expense are likely to limit operational use of this technique on large scale outbreaks.

Attractants have the potential of improving detection of major outbreaks and of creating a sink for breeding insects during incipient stages of epidemic development (Rudinsky, 1963). The attractants are chemical pheromones that are set out in stations distributed throughout potential epidemic areas. Amounts of chemical are small, and the chemicals used are not regarded as pesticides. Operational utility of this concept is regarded as promising but no major outbreaks have been controlled at this time on the basis of its use exclusively.

Silvicultural control of insects is a time-honored system that has several major shortcomings in the treatment of present-day epidemics. This method involves the maintenance of stands in a state of composition, stocking and vigor so that insect populations remain within endemic levels. The same procedures that are used for management, i.e., thinning, composition control and high vigor management, are also normally effective means of improving the economic outlook for the stands themselves.

Limitations of the silvicultural methods are inherent in the nature of the resource to which they need to be applied. This approach requires the intensive management of extensive contiguous areas of forest. Epidemics of the major defoliating insects are often focussed in areas that are largely inaccessible, or at least impossible to place under management quickly. Outbreaks of the Douglas-fir tussock moth and spruce budworm have a tendency to occur in extensively managed stands, or those in which no management has been initiated. In the former, particularly, the silvicultural requirements for avoiding outbreaks are only now under investigation. In the latter, there is evidence that composition favoring spruce as a greater component than true firs, and maintenance of all stands in a vigorous growing condition minimizes budworm damage. In much of the spruce-fir forest region, reserves of overmature stands (in reference to budworm activity) are great enough so that many years will elapse before young, managed stands dominate the region. The scale of operation needed to implement silvicultural control on the eve of an impending epidemic reduces its value for short-term use. In the long run outlook, however, silvicultural management of stands vulnerable to defoliators can be expected to reduce outbreak scale and intensity.

DAMAGE BY RODENTS TO DESIRABLE TREE SPECIES

Rodents and other wildlife cause major damage to forest regeneration. Methods used to minimize losses include direct control of animals with rodenticides, protection of seedlings with physical barriers, use of repellants, vegetation management to control animal habitat and the planting of species of low vulnerability to animal damage.

Rodenticides include strychnine alkaloid, endrin and zinc phosphide. Of these, only endrin is applied by aircraft, and only as a seed treatment. Sodium fluoroacetate (compound 1080) was used for this purpose, but is no longer registered.

Rodenticides are prepared in baits. They are distributed so that target animals will be attracted without causing major harm to nontarget species. Quantities of toxicants are extremely small. The most widespread of these is endrin, used for treating conifer seed in aerial seeding operations in the Pacific Northwest. Endrin is applied to seeds in a concentration of .5 percent of seed weight. Treated seeds are distributed at the rate of one-half to one pound per acre, with attendant distribution of endrin at the rate of .0025 to .005 pound per acre, (3 to 6 grams per hectare). Endrin has been recovered from streams after such applications at a maximum concentration of 0.00006 mg/l, decreasing to the detection limit of 0.000001 mg/l in 10 days (Moore *et al.*, 1974). Because of high risk of failure, aerial seeding is losing popularity, and the use of endrin has decreased. Other rodenticides are not applied broadcast, and have no direct access to water supplies. In general, aerially applied rodenticides do not constitute an appreciable toxic hazard to aquatic systems or water supplies. Their future use will likely be restricted to reforestation of burns for which planting stock is not available, and inaccessible steep slopes.

Physical barriers to protect seedlings have come into large scale use in the Pacific Northwest. Plastic tubing and stakes are placed on each seedling after planting. Large rodents, deer and elk are prevented from causing mortal injury to seedlings. Costs per acre for establishment and protection are very high, but this method has gained acceptance because of its dependability, and the wide range of animals from which it offers protection

Chemical repellants include a zinc complex (ZIP), tetramethylthiuramdisulfide (TMTD) and a recent development of a sulphurous compound made from eggs (BGR). These materials may be applied to seedlings in the field or nursery by locally applied ground sprays, and do not represent a hazard to water quality. TMTD and BGR have been applied from aircraft in research studies, but with varying effectiveness in preventing animal damage.

Control of animal damage has been demonstrated in several studies of habitat control. Borrecco, 1973; and Borrecco *et al*., 1974; have shown that vegetation management with herbicides can have a marked temporary influence on the use of a regeneration area by certain wildlife. Herbicides specific for grasses and broadleaf weeds each had a role in the modification of habitat, and the effect on animal use was related to duration of preferred species control. This approach is clearly not a non-chemical method, but substitutes chemicals with tolerances established in human food in place of highly toxic rodenticides. The method is ecologically logical because of its reduction of carrying capacity for offending animals. The herbicides involved, atrazine, simazine, dalapon and 2,4-D relate to water quality as outlined under the discussion of moisture dificiencies attributable to herbaceous weeds. Treatments for animal damage control also provide seedling growth response, reducing vulnerability to future injury.

CRITERIA FOR LIMITING CONCENTRATIONS OF CHEMICALS IN WATER

According to the Title I amendments to the Federal Water Pollution Control Act, Sec. 101, the national goals relating to pollution sources are repeated:

1) that the discharge of pollutants in the navigable waters be eliminated by 1985.

2) that, wherever attainable by July 1, 1983, an interim goal of water quality be achieved so as to provide for the protection and propagation of fish, shellfish and wildlife, and provide for recreation in and on the water.

3) a national policy was also established that the discharge of toxic pollutants in toxic amounts be prohibited.

The purpose of this particular set of guidelines for pollution control is to enable conformance with these goals and this policy by eliminating harmful effluent arising from non-point sources related to use of silvicultural chemicals. Of critical importance in the achievement of this objective is the identification of those harmful effects to be avoided. Once harmful effects are perceived, target levels of water quality can be established below which social and biological systems are undamaged.

Because the non-use of all silvicultural chemicals is not in the best interests of the nation's forest resources, maximum concentration limits must be set for chemicals in water that insure both safety and conformance with the national goals and policy. And they must not force land managers to resort to demonstrably harmful substitute practices. That is, the cleanest possible water and good forest management are both desirable. The purpose of this section is to propose 1) a system of classifying streams and use patterns in relation to water quality objectives, and 2) within this approach, express concentrations and conditions representing upper limits of allowable concentrations while defining where they are tolerable and under what circumstances stricter standards are necessary. These levels are not intended as guides for irrigation systems and impoundments in which higher levels of herbicide may be required for aquatic weed control.

Maximum concentrations for some of the silvicultural chemicals are listed in the 1975 draft edition of Quality Criteria for Water (QCW), proposed by the Environmental Protection Agency. Several of the most important chemicals used in silviculture, including the most controversial, 2,4,5-T, are not discussed, however. The chemicals that are mentioned are assigned single numbers reflecting an upper limit of concentration, regardless of the system in which the pollution is occurring.

This document proposes an expanded treatment for certain chemical criteria in relation to forested watersheds, and offers criteria for all silvicultural chemicals, including some covered by QCW. The group of criteria proposed here contains allowances for variation in stream size and use. The approach combines biological and statistical concepts to estimate thresholds of harm to ecosystems. These estimates are based on the nature of the ecosystem, the substances to which organisms in the system are exposed, and the form, magnitude, and duration of exposure. The assumptions and value judgements are given.

A listing of recommended tolerance levels, and conditions under which tolerances vary, is given for all registered pesticides and fertilizers likely to be applied so as to reach forested waters as the result of silvicultural use. Potential synergism among chemicals is not considered.

Silvicultural chemicals are found at very low levels in streamwater, and two or more chemicals are seldom used simultaneously. A revision of this approach may become necessary as intensity of forest management increases. A brief explanation of toxicological principles and of the data on each chemical will follow.

Relation Between Dosage and Response

Exposure of organisms to toxic substances results in the uptake of some of the substance. The amount taken up in relation to body weight determines the degree of effect in a quantitative relationship. This relation may be linear or non-linear, depending on mode of action.

Acutely toxic substances have more of an effect when a given amount is administered as a single dose than when divided into numerous small doses over a period of time. Mortality is a standard index of response. Chronically toxic materials act in the reverse manner, and cumulative effects on behavior, reproduction or oncogenesis are some indicators of exposure. Sublethal effects of chronic exposure may be investigated in terms of reactions of eggs or atypical reactions of fish to their physical environments (Muirhead-Thomson, 1971).

The relation between dosage and response is expressed differently in the above two classes of chemical. Acutely toxic chemicals characteristically produce a log-linear dosage response within a finite range. Below this range, the organisms deactivate the chemical without permanent injury; above this range, all individuals are dead and no further response can be tested. Chronically toxic materials do not follow the same pattern. Low levels of continuous exposure result in cumulative deposits in vital processes, leading to manifestation of long-term difficulties in general metabolism, reproduction, or tumor initiation. Chronic exposure to acute toxicants at subacute levels can elicit a chronic reaction, usually reversible when exposure ceases. Thus, both exposure rate and duration of exposure are important in evaluation of toxic hazard. The former tends to be of paramount importance with acutely toxic chemicals, the latter with chronically toxic substances.

The term "acutely toxic" should be distinguished from "highly toxic". The former refers to a type of toxic action, while the latter refers to degree of toxicity. Most herbicides used in forestry are of low to medium toxicity, and are of the acutely toxic group. The insecticides other than chloroniated hydrocarbons are more variable in mammalian toxicity, but are still in the acutely toxic category. The chlorinated hydrocarbons may be either of the acutely toxic or chronically toxic class, and also have a wide range of mammalian toxicity.

The "No-Effect" Level of Toxic Chemicals

Direct harm to organisms results from exposure to toxic chemicals only when sufficient toxin enters biochemical systems to interfere with critical functions, i.e., the "threshold" dosage is exceeded. Despite the extreme toxicities of some chemicals, harm is not likely to occur when the first molecule of toxin enters the organism. The reason for this is that cells are equipped with many functional molecules at each link within each metabolic pathway. For many metabolic functions, there are several pathways and many sites within each pathway that can keep the organism functioning when one biochemical system or site is temporarily blocked (Loomis, 1974).

Furthermore, all toxicants are sorbed, inactivated or detoxified to some extent by metabolic action, and surviving organisms usually emerge from sublethal exposure to acutely toxic substances without permanent injury or harmful residue. This expectation is valid for most chemicals used in forest operations, because of their low toxicity to animals and moderate to rapid decomposition. This generalization does not hold for chronically toxic materials, of which none except rodenticides are registered for aerial application in forests.

Demonstration of precise "no-effect" levels of toxic exposure poses social, biological and statistical difficulties. It can always be argued that humans remain untested, hence are subject to uncertainties resulting from testing with other mammals. This uncertainty will remain a social problem in administration of any allowable-level guidelines, and is the basis for using wide safety margins when extrapolating from tests to regulations. Resolution of this problem is beyond the scope of these guidelines, but some guides for safety factors are presented below for inspection.

In the biological estimation of "no-effect" levels, it is necessary to use measures other than traditional acute toxicity tests to determine exposure rates leading to weight loss and other symptoms. In standard tests, mammalian toxicity is determined according to a median lethal dose (LD_{50}); fish sensitivity is gauged by a median lethal concentration in water (LC_{50}).

The variation among species, and among individuals within species, are often examined for acute toxicity with traditional statistical methods for predictive purposes. Typically, symptoms within a test population of organisms show up according to a cumulative frequency distribution as in Figure 1. The distribution of responses illustrated in Figure 1 is a translation

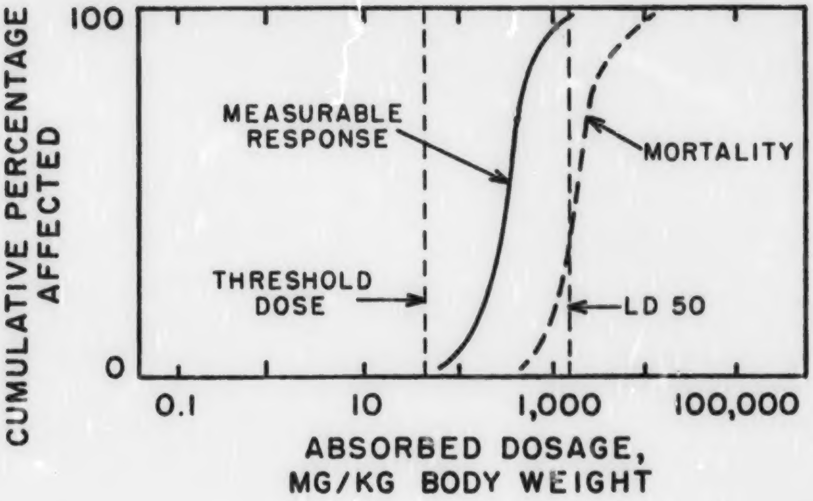

Figure 1. Typical dosage - response curve for an animal population fed a toxic substance. Note that dosage is based on units of toxicant per unit of body weight. Threshold (no effect level) is the dosage below which organisms detoxify chemical as fast as it is absorbed. These curves are transformed normal distributions. In laboratory tests, slopes of curves vary among species (Muirhead-Thomson, 1971).

of a normal distribution of incremental responses to incremental dosages of toxic substances, (Figure 2). In conducting tests for population responses, samples of variable individuals are used, leading to an assumed bell-shaped normal distribution. Increasing the number of samples can increase the precision of estimate of median response point, but cannot identify either "no-effect" or "100-percent effect" levels exactly, regardless of sample size (Figure 2).

The traditional use of the normal distribution in tests of acute toxicants has been unsatisfactory. Because a normal distribution cannot conceptually show a zero expectation of responses, an infinite number of samples still cannot predict a zero response level even when one is known to exist. The best it can do is to predict a very low probability of observing a response, even at a zero dosage. But zero or undetectable responses appartently do occur, as attested by frequent references to good health of humans and animals exposed to low levels of an imposing array of natural and synthetic toxic materials.

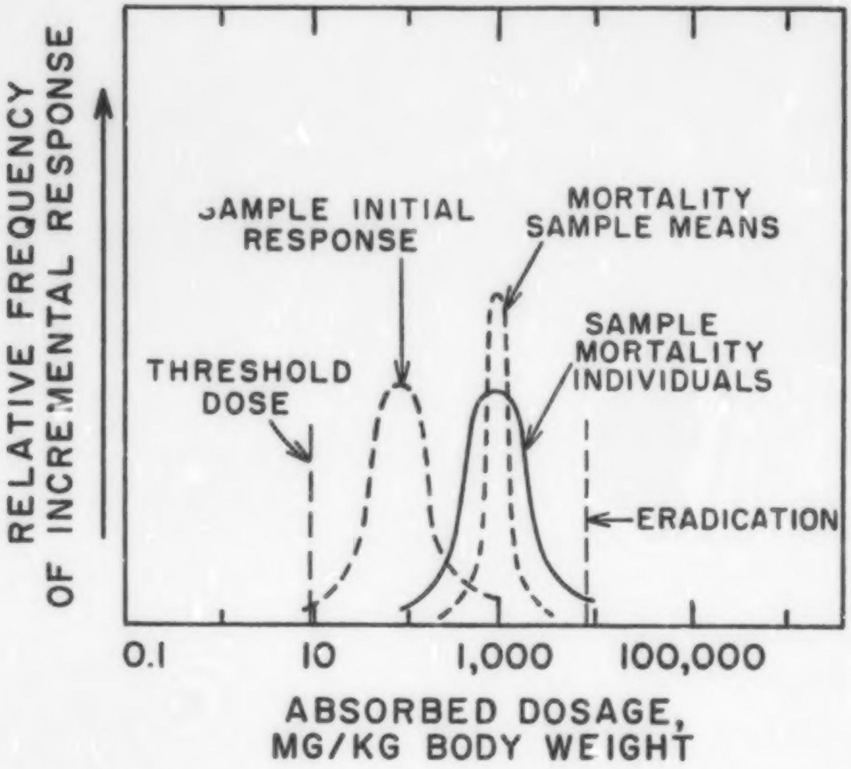

Figure 2. Normal distribution of sample responses to incremental levels of exposure to toxic substances, and distribution of estimated sample means based on the sample.

Experience has shown that biological systems are remarkably well buffered against toxic agents. Many natural foods contain toxins that are tolerated daily at low dosages, but without observable adverse effects; indeed, some natural materials that are toxic at high dosages are required for adequate nutrition at some lower dose. Examples include copper, various vitamines, sodium chloride and many others. The synthetic toxicants similarly appear to have "no-effect" levels, even though the assumption of a normal distribution of responses would preclude this conclusion.

The normal distribution similarly fails to predict threshold responses of numbers of species within mixed species communities. Forest species may be classified into several broad groups for convenient discussion of generalized community responses. Higher plants, fungi, fish, birds and mammals, and insects differ substantially in their absolute sensitivity to pesticides and in their exposure to aerially applied sprays. In general, plants and insects receive a heavy exposure in relation to total metabolic tissue because of high surface-to-volume ratios, surface orientation or both. Conversely, mammals have low surface-to-volume ratios and absorb poorly through skin, while fish having similar surface area are further exposed through gill action. Absorptive surfaces of fungi are usually protected from direct exposure. There is therefore a generalized physical selectivity among organisms based on variable likelihood of exposure to, and uptake of, a given absolute dose. Superimposed on this exposure variable is a wide range in sensitivity. If the absolute toxicity of a pesticide, eg. PGBE ester of 2,4-D, were considered in relation to these groups of organisms, it is likely that responses would appear as in Figure 3a. The observed responses to variable rates of application, however, suggest that specific toxicity is a poor estimator of hazard in broadcast applications (Figure 3b).

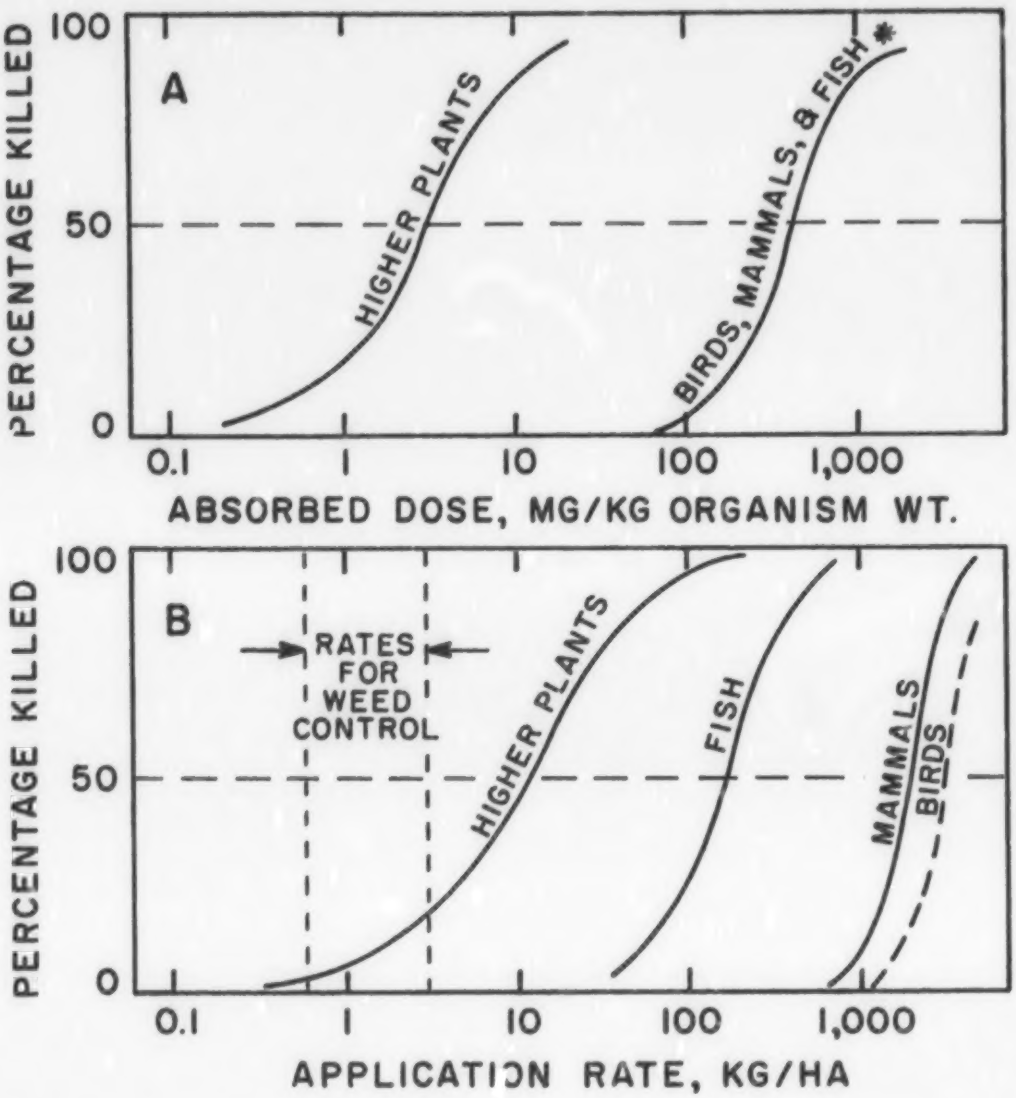

Figure 3. Responses of several groups of forest organisms to 2,4-D PGBE ester in toxicity tests and in projected field applications, without stream protection.

*NOTE: Insufficient data are available to illustrate responses of insects. Fish data assume 100:1 concentration factor from water concentrations in LC_{50} studies, and 0.01 mg/l in streamwater per kilogram-hectare application rate. Mammal data assume consumption of five percent of body weight in forage. Field data from birds inadequate for generalized response. Dotted line assumes seed and insect foods contain 50 percent of that found in herbaceous forage, 20 percent of body weight consumed per day. Mammalian responses in field assume contamination of forage of 5 mg/kg for each kilogram-hectare application rate and consumption of five percent of body weight per day.

Adequate data are not available to plot responses of forest fungi. The fungi are important in water quality as the result of their roles in retention of nutrients and other solutes. In several reviews of impacts of agricultural pesticides on soil microflora, adverse effects were not reported at field use rates (Audus, 1970; Bollen, 1961, 1962).

The expense of conducting "zero-response" tests on many species would preclude precise identification of zero asymptotes even if such were possible. Safety factors are used to allow for variation in both sensitivity among individuals and in the rate of exposure. Extra allowances are made for safety as insurance against spills and species of unknown sensitivity. The knowledge that species vary in response to toxins leads to the assumption that there is always a more sensitive test species on which effects haven't been evaluated, perhaps man. But groups of species tend to run in patterns. Barring allergenic reactions, classical screening will identify the groups illustrating extreme sensitivity. When these groups or members thereof are identified as pests, the chemical is registered as a pesticide for their control. Typically, registered herbicides are poor insecticides, fungicides or rodenticides; insecticides are ineffective on plants but may be highly toxic to other animal life. Traditionally, it has been uneconomical to use chemicals for controlling species other than the most sensitive. The pesticides registered for use in forestry mostly have undergone extensive use and testing in agricultural crops. The likelihood of major unexpected sensitivity appears remote.

Statistical problems are nearly insurmountable when extrapolating toxicity data from limited test populations to a virtually unlimited human population. Scientists traditionally ignore probabilities of 1:1,000,000 in test animals; they are untestable for practical purposes. Yet the expectation of harm to one human in a million is unacceptable. Mantel and associates (1975) have developed probit-based testing procedures leading to extrapolation for populations. These procedures have been used and standardized for chronic carcinogenic and teratogenic compounds, largely pharmaceuticals. They have not been associated with toxicity data regarding silvicultural chemicals, and are probably inappropriate for most acute toxicants because of their dependence on continuous normal distributions of responses.

For the present, the assumption must be made that there is indeed a "no effect" level of every toxic chemical that does not produce an allergenic reaction. Support for this assumption is widespread (CAST, 1974) but there are many who object. The assumption is accepted here because the objections have been identified principally with respect to chemicals other than those used in forestry, with emphasis on chronic toxicants, carcinogens and teratogens. It must remain an assumption, however, because of the conceptual impossibility of extending the normal curve to zero response. The chemicals other than chlorinated hydrocarbons registered for use in forestry are not chronically toxic except as noted hereafter, nor are they carcinogenic or teratogenic at field use exposure rates. Several of these chemicals have produced terata and tumors at extreme exposure rates, and such occurrences enter into the establishment of safety factors.

The no-effect level is not the same for all organisms. It is necessary to understand which organisms are likely to be exposed and how they are exposed in setting concentration standards, and to make value judgements as to which organisms or groups of organisms may be treated with some risk.

Fish, aquatic plants, benthic organisms and water users are considered in this analysis to be the most vulnerable to toxic hazard. Unfortunately, the same sets of test data are not available for each group of organisms exposed to each chemical. It is therefore necessary to establish patterns of activity within chemical and test animal groups. While extrapolating from groups to individual species increases the uncertainty factor, a systematic examination of patterns of activity of the chemicals, and of the place in the food chain of the most sensitive organisms, helps to develop a perspective of the responses of individuals within groups. These will be described later in the rationale for setting concentration criteria.

The Roles of Concentration and Exposure in Estimating Toxic Hazard

Concentration of a toxicant is only one index of potential harm to an exposed organism. Toxic hazard is related to level of exposure, duration of exposure, route of contact with the organism, and absolute toxicity and chronicity of the compound. A single statement of concentration setting upper limits on contamination is not adequate as a measure of harm prevention.

Concentration of a toxicant directly affects aquatic organisms only in relation to the amount absorbed and period of retention. The presence of a chemical is of no direct consequence unless the chemical enters sensitive metabolic systems. This can occur through skin absorption, ingestion or respiration; ingested material can enter either as a water solution or as contaminated food or particulates. The likelihood of a given chemical concentration in water or food producing a specific body burden depends on its ability to penetrate, persistence, (duration of exposure) retention in the organism and partitioning of the toxicant between water and food or suspended solids. Also important is the ability of the chemical to be absorbed through skin and intestinal membranes. Chemicals with a strong affinity for surfaces may adhere to outer surfaces of food sources for other organisms.

Virtually all literature reporting responses of fish to toxic chemicals expresses concentrations in terms of lethal concentrations to 50 percent of a population (LC_{50}) in 24, 48 or 96 hours. Some experiments have been run to determine weight gain or loss and other subacute effects, but very few studies are conducted in streamlike environments. Aquarium tank studies do not give an accurate indication of lethality in streams. The nature of their bias tends to overestimate expected toxicity in wild systems (Muirhead-Thomson, 1971), thereby providing some safety factor at the outset. They also protect fish from predation and certain diseases, however, so that there is some compensation of bias.

Aquarium studies lead to overestimates of toxic hazard in several important respects. The first is that they test the effects of a known concentration that is not diluted over a period of time as a stream would be; the second is that a glass system does not have substantial adsorptive capacity to tie up solutes. A third problem is that streams tend to offer some escape opportunities downstream for mobile organisms. Steady states of elevated toxicant concentrations do not occur in forest streams as a result of the use of silvicultural chemicals. Forest watersheds are characteristically treated with chemicals that are immobile in soil. Chemicals generally enter water as droplets that drift onto open water surfaces, generating contamination pulses or "spikes" that begin to dissipate immediately (Norris, 1967). The integral of concentration during 24 hours is less for a variable concentration with a given peak than for a steady state at the same maximum concentration (Figure 4). Assuming that such an integral is a reasonable index of exposure in water, the spike concentration pattern constitutes a lesser hazard to aquatics than a steady state of comparable maximum. The tightly adsorbed character of most of the pesticidal chemicals, together with the presence of suspended soils and rough bottom surfaces, removes the chemicals even more rapidly than would be expected because of downstream movement alone (Norris and Montgomery, 1975). Retention of chemicals by adsorption raises several questions, however. Adsorption retains the chemicals so as to extend the potential exposure period, albeit at low levels. For

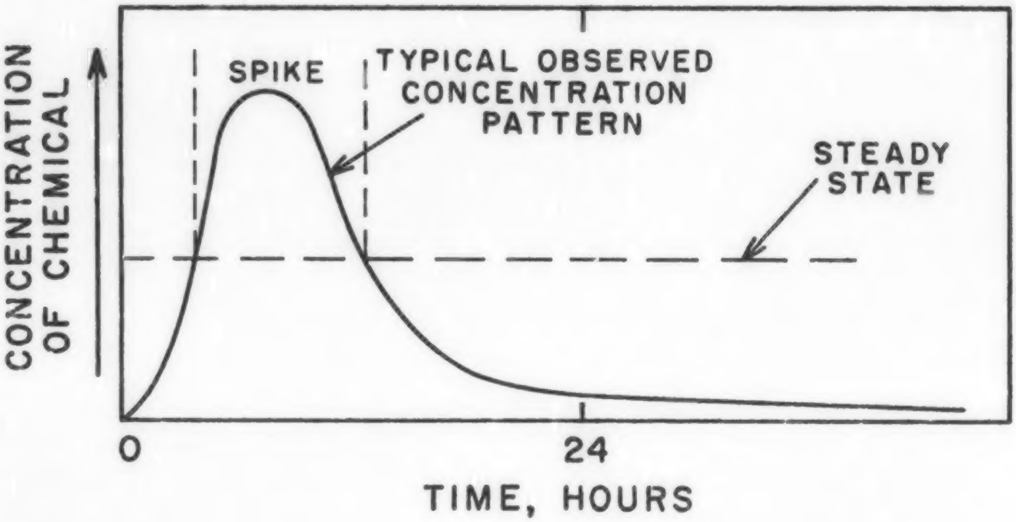

Figure 4. Comparison between typical pattern of concentration of chemical in water from aerial application compared to static concentration (such as in aquarium test) leading to same total exposure during 24 hr. period. Note that the maximum concentration is higher in the field, but that the subsequent exposure is lower.

chronically toxic materials, this phenomenon could increase toxic hazard. Degradation occurs in the adsorbed state, and the possibility exists that toxic metabolites could be released within the aquatic community. Although this may occur, the small amounts of chemical retained and lower level of total metabolite than parent compound reduces the expectation of toxic hazard to considerably lower than that of the original contamination. Retention could conceivably lead to reaction with stream nitrate, leading to formation of N-nitroso derivatives of oncogenic potential. This process is concentration dependent, however, and the concentrations of both pesticides and nitrate are extremely low for such to occur. Rapid dilution of chemicals occurs as water moves downstream from forest operations, reducing peak concentration and perhaps providing escape opportunity for larger fish that would not be predicted from aquarium tests. It therefore appears reasonable that the estimates of "no-effect" levels from aquarium tests would hold for wild aquatic systems, but that the reverse would not necessarily be true.

The presence of synergists can modify pesticide behavior in relation to a particular organism. Muirhead-Thomson (1971) observed that detergents and herbicides can influence sensitivity of fish to insecticides. This can occur either through a slight change in the health of the organism that increases vulnerability to other toxins or disease, or it can change the organism's health or habitat so that it is subject to other harmful influences. Estimation of direct effects in the presence of such confounding factors decreases feasibility of field studies to support extrapolation from aquarium tests. The tank tests are therefore necessary, but require a judgement-based safety factor for field use in the absence of validation.

The final point in estimating toxic hazard is the identification of chronic effects from long term exposure. "Chronicity" is defined as the ratio of dosage necessary to produce some level of response when administered to laboratory animals as one dose, as opposed to many daily doses, typically over a 90-day period. The number identifying chronicity is a measure of the importance of minimizing any long-term exposure to a compound; the lower the chronicity index, the greater the assurance that harm can be avoided by avoiding peaks in the demonstrably toxic range of concentration. Hayes (1967) derived chronicity indices for an anticoagulant rodenticide (Warfarin), a chlorinated hydrocarbon (DDT) and several organophosphates (dichlorous, parathion and guthion). He observed very high chronicity values (20.8) for the warfarin, intermediate for DDT (5.6) and low for the organophosphates (0.8-1.16). He defined as those with a chronicity ≤ 2.0 to be acutely toxic.

Fortunately, nearly all pesticides registered for use in forestry are of the acutely toxic group. Their rapid disappearance in aquatic systems permits dependence on regulation of peak concentrations for insurance of fish safety. Endosulfan and lindane, seldom used near water, are the only persistent and potentially chronically toxic compounds in use; reports regarding their theoretical chronicity were not found in this review. They have not been reported as major water contaminants after forest insect control operations.

Application Factors Used in Setting Water Quality Targets

A "safety factor" is a ratio used to extrapolate from the lowest observed toxic doses to estimated "safe" levels. It is not necessary or economically attractive to use chemicals at rates that cause serious problems in the usual course of operations; there is no justification for contaminating at lethal levels. However, it is necessary to use some chemicals, and some quantity may inevitably reach some body of water. Below a certain level of use, the consequences of reduced usage inevitably lead to an increase in some other adverse consequences, such as lowered forest productivity or an increase in use of a more damaging practice. Thus, in the attempt to minimize total harm to ecosystems, it is necessary to allow the temporary existence of low levels of contamination, with tolerable levels established at some percentage of those estimated to cause "negligible" effects. Concentrations producing negligible effects cannot be determined readily in the field, and "application" factors are used in estimating them from laboratory data regarding lethal dosage. These are needed to compensate for the admittedly imprecise lab data in setting water quality criteria that provide safety over a range of conditions and species.

Forest watersheds have some features in common that influence the expectation of major harm from delivery of a toxic chemical into a stream. Application factors can vary in different situations without compromising safety. If a forest stream is small, it is likely that it will sustain elevated concentrations for a matter of hours, and will join a larger stream shortly. If fish are present, the population may be exposed without injury to concentrations of chemicals that would cause injury over a prolonged exposure period. The short term of exposure and escape opportunity offer more safety than an aquarium test, and application factors can be relatively small. If the stream is of large size, a given input of chemical will be substantially diluted at the time of infall, and the expectation of local harm will be less than for the smaller stream although the ability to recover may differ up or downstream. Contamination of the larger stream at a harmful concentration poses a much greater chance of causing harm downstream, however, because of the greater duration of exposure. Therefore a larger factor is needed when larger streams are involved. Even in an aquarium, a factor of 5:1 from LC_{50} levels reduces mortality to low levels with some acutely toxic materials (Muirhead-Thomson, 1971). Larger factors are needed in progressively larger aquatic systems to provide increased insurance. Nevertheless, feeder streams are extremely important habitat for certain organisms, and contamination of large numbers of feeder streams increases the contamination of large streams. Because of the greater likelihood of massive applications, and of direct injury to aquatic organisms, insecticides require larger factors than herbicides as a general principle.

The above has outlined some generalizations that can be summarized as follows:

a) For acutely toxic materials, limiting peak concentrations in streams on the basis of aquarium tests of 24 hours or more offers more margin of safety than is suggested by aquarium data.

b) For chronically toxic materials, large safety or application factors are needed to insure against long-term effects. The basis should be a longer-term exposure test, in the absence of which a 10-fold increase in application factor is a minimum.

c) For any toxic material, small forest streams cleanse more rapidly, hence will tolerate a given peak level of contamination more readily without major harm than will larger streams. There is therefore justification in using smaller safety factors on small streams than on large ones, except where spawning streams are treated at critical seasons. At such times, factors for the next larger order of stream should be used.

d) Lower maxima are appropriate for extended exposure than for short-term peaks with the same application factors in use. Since large streams cleanse more slowly than small streams, and more diverse communities are exposed, the larger the stream, the larger the needed application factor from aquarium tests.

Proposed Criteria for Water Concentrations in Forest Watersheds

Criteria are given in Table 3 as targets toward which management practices can be directed. Rationale for each proposed concentration have been developed from a review of literature on a variety of test organisms. Each chemical for which an allowable concentration has been set has been evaluated either as a natural component of aquatic systems and human diet, (e.g. arsenic) or as a synthetic toxicant administered to a variety of test animals including fish. Many were tested on aquatic insects and crustaceans.

All proposed target maxima in feeder streams are substantially below the concentrations causing significant mortality in the most sensitive known test organisms. When the most sensitive test organisms were either resilient species of insects or of crustacean species seldom inhabiting headwaters of the forest streams, and data reflected 48-hour or more exposures to acutely toxic chemicals, application factors of 20:1 below LC_{50} concentrations were used because of the pulse nature of contamination. Greater application factors were used where fish were the most sensitive group of organisms (100:1), and for chemicals of suspected chronicity or known to accumulate in biological systems (200:1) unless otherwise noted. These factors are increased for larger streams.

Concentration limits reflect the use to which water will be put. Among the group of chemicals under consideration, e.g., some nitrates, are of principal concern to the health of humans. Some are of concern because of their immediate effect on fish; some of the insecticides are in this class, especially chlorinated hydrocarbons. The herbicides, in particular, are of concern more in potential irrigation water quality than in relation to effects on fauna. Where water is not used for irrigation, criteria may be based largely on hazard to aquatic species and humans.

The final point in qualifying target concentrations was stream magnitude. Small streams, i.e., those with flow rates less than 10 cubic feet per second (cfs) elute quickly and are rapidly self cleansed after a pulse-type contamination. There is no opportunity for sustained exposure, and organisms not acutely influenced by the short term peak of chemical concentration can be expected to recover rapidly and normally after exposure to the chemicals listed. Larger streams and rivers, even of fast-moving water, elute more slowly because of integrating much larger potential treatment areas. Other important reasons for decreasing the concentrations allowable in intermediate and larger streams are:

1) Concentrations change slowly in large or slow-moving streams, and a greater chance for chronic exposure occurs from a given concentration.

2) Large streams at elevated concentrations elute larger amounts of chemical into the next larger stream than feeder streams of the same concentration, thus creating more general pollution problems for a given level of contamination.

3) Large streams contaminated at high levels offer little escape opportunity for organisms.

4) Sustained high levels of contamination, as might be associated with pollution of larger rivers, offer maximum opportunity for biomagnification of compounds having this tendency.

5) Large streams have a high probability of being used for domestic or irrigation use close to the point of contamination.

Accordingly, the following rationale are offered in support of concentrations suggested as targets in streams of the various magnitudes. Specific references and data citations are summarized in Appendix Table 1.

NITRATE

Nitrate is a highly soluble oxidation product of nitrogen that may be an end product of any nitrogen fertilizer. Its natural occurrence in soil and water is ubiquitous. The criterion of 10 mg/1 (as nitrogen) for all streams is the same as recommended by the QCW. The rationale is that the principal recipient of harmful effects from a short term elevated level could be human infants. No distinction is made between stream size or duration of exposure, because nitrate is a chemical found naturally, and peak concentrations can be superimposed on high natural levels so as to cause difficulties. The same guide is used for irrigation water because of its effect on ground water. Because of the pulse nature of potential contamination, potential effects on eutrophication are not considered here.

The point at which nitrate should be monitored should be at the source of potable water, or at the irrigation pump.

PHOSPHATE

Phosphate is an oxidation product of phosphorus, and is the form found in rock phosphate fertilizer. Phosphoric acid is also used because of greater solubility. Due to the wide range in phosphate concentrations reported from natural forest waters, no criterion is suggested here. This principle is in agreement with that followed by the Quality Criteria for Water.

Data currently available from the Atlantic and Gulf coastal plain area, do not indicate significant water quality effects from phosphate fertilization at present. However, preliminary results (Avers and Scott, 1976) suggest that if large portions (greater than 25%) of watersheds with very shallow water tables receive heavy applications of phosphate, slightly increased phosphate concentrations in drainage water can be expected. The effect of these levels on potential eutrophication is dependent on several factors including temperature and turnover rate of impounded water. Economics of the timber industry in the Southern U.S. indicates that management will become more intensive and acreage under intensive management will continue to increase. Periodic investigation of water quality effects from forest fertilization in this area are thus advised, and future guidelines on water concentration may prove feasible. These criteria should be established to reflect the properties of the watershed in the maximum permissible levels.

AMITROLE (3-amino-1,2,4-triazole; amino triazole)

Amitrole is a water-soluble compound used in the Pacific Northwest for control of salmonberry in reforestation. Concentrations of this herbicide are not considered in the QCW. Amitrole is a low-toxicity compound, with acute oral LD_{50} in rats of 1100 mg/kg, and LC_{50} (48 hrs.) for coho salmon of 325 mg/l. The two principal reasons for establishing a large safety factor are that amitrole has been identified as a weak carcinogen (Jukes and Schaffer, 1960), and its application in irrigation water could conceivably cause crop injury. Because of amitrole's status as a carcinogen, an extremely large safety factor is needed to insure minimal risk.

The data for estimation of carcinogenicity of amitrole is not available, and it is not possible here to estimate dose-response slope. Mantel's procedure (Mantel _et_ _al_., 1975) is useful in identifying the need for large safety factors when dealing with any carcinogens. Amitrole's status as a carcinogen is open to doubt because of reportedly reversible effects at 500 ppm in rat feed for 17 weeks (Jukes and Schaffer, 1960). Because concentrations are removed rapidly from water by stream bottom complexing and other causes, (Marston _et_ _al_., 1966; Norris _et_ _al_., 1966) the concern with extended persistence at elevated concentrations is moderated. Under the circumstances, application factors for potable water are based on _Daphnia_ response, at 200:1 for extended exposure periods and larger streams, 100:1 for those streams of medium size and 20:1 for small feeder streams. This provides for maxima of .015 - .15 mg/l in potable water. For irrigation water, an upper limit of 0.1 mg/l is proposed to minimize potential damage to crops. The potable water criteria are established on the basis of known responses by crustaceans, rather than mammals which are generally substantially less affected by amitrole. Mammalian safety should therefore be well protected by the given application factors, with margins of greater than 500:1 for short term exposure (Jukes and Schaffer, 1960) and 5,000:1 in the longer term.

AMMONIUM ETHYL CARBAMOYL PHOSPHONATE (Krenite)

This herbicide is used to control several deciduous brush species during reforestation. The water-soluble material has unique properties relative to both brush killing mode of action and activity in water. The material is a growth regulator related to both the carbamates and phosphates while exhibiting the properties of neither. Mode of action is apparently specific to inhibition of elongating cell walls, although there is limited data on this subject. Among the few organisms tested, toxicity is neglibible to all fauna. Because there is little published data, Dr. James Harrod, Biologist, DuPont Co., Wilmington, Del., was consulted for unpublished findings. He indicated that Krenite has been applied to boll weevils, aphids, roaches with no effect at any dosage including 0.5% active drenching sprays. At the highest dosage, carpet beetles and flies demonstrated some injury but low mortality.

Bioaccumulation has not been observed in aquarium tests; residues in fish were comparable to concentration in water. Teratological tests were negative; carcinogenesis tests have not been done. Soil residue is short, with half of the herbicide converting to carbamoyl phosphonic acid in two weeks, which is oxidized to CO_2 and humic acid fractions within 8 weeks. The highly soluble material adsorbs rapidly onto soil particles, and is not taken up by roots or fungi. Disappearance from bottom sediments occurs over a period of three months or less.

Ammonium ethyl carbamoyl phosphonate is evidently a very low hazard herbicide. The rationale for setting maximum levels in rivers of 0.5 mg/l is that the 1,000:1 application factor, based on bluegills, allows use for all riparian brush control, and still provides for a large margin of safety in consideration of the largely unknown properties of toxicology. Smaller streams permit higher levels, as shown in Table 3, on the basis that more is known about acute toxicity, and a wide safety margin is perceived even when allowing concentrations consistent with aerial application to open water.

ARSENICALS (organic)

Cacodylic acid and MSMA are naturally occurring pentavalent intermediates in the natural global arsenic cycle. They are used largely for tree injection for control of unwanted trees, and also for control of bark beetles. They are not considered in the QCW, which addresses arsenic as an elemental pollutant. Because the pentavalent arsenicals are metabolized to form low level equilibrium concentrations of trivalent compounds of greater toxicity, it is appropriate to consider toxicological data from the trivalent materials in relation to prolonged exposures. It is also germane that a natural background of elemental arsenic is ever present, and forest ecosystems have equilibrated harmoniously in the presence of natural concentrations of arsenic as As_2O_5 ranging up to 6 ppm in soil (Greaves, 1974; Norris, 1974) and that benthic organisms maintain populations at bottom mud concentrations of 1,920 ug/g arsenic (Leuschow, 1964).

The levels of arsenic concentration suggested by the QCW were based entirely on inorganic trivalent arsenic, and were set at 0.05 mg/l for potable water and 0.10 mg/l for irrigation water. It is recommended that these be accepted for a silvicultural target, with the exception that an increase to 0.10 mg/l is recommended for smaller feeder streams as well, provided

Table 3. Recommended concentration maxima for silvicultural chemicals by stream class and user group. Potable waters include safety factors for wildlife and aquatic organisms as well as humans.

				Criteria, PPM 24 hr. Mean Stream Class & User					
				< 10 cfs		10 cfs-Navigable		Navigable	
Class	Chemical	Most Sensitive Test Species Affected	Test Basis & Concentration	Potable	Irrig.	Potable	Irrig.	Potable	Irrig.
Fertilizer	Nitrate	Man	No effect, 10 mg/l N	10*	10*	10*	10*	10*	10*
	Phosphate	Algae	Growth response var.	-------inadequate basis for recommendation---------					
Herbicide	Amitrole	Daphnia	LC_{50} 48 hr, 3 mg/l	.15	0.1	.03	.01	.015	.01
	Ammonium ethyl carbamoyl phosphonate	Bluegill	LC_{50} 48 hr, 670 mg/l	5	5	1	1	0.5	0.5
	Arsenicals (organic)	Man	No effect, 0.12 mg/l	.1	.1*	.05*	.1*	.05*	.1*
	Dalapon	Daphnia	LC_{50} 48 hr, 11.0 mg/l	.5	.1	.1	.02	.10	.02
	Dicamba	Bluegill	LC_{50} 96 hr, 23. mg/l	.2	.004	.05	.002	.01	.001
	Dinoseb	---------------------inadequate data---------------------							
	Picloram	Bass	LC_{50} 48 hr, 19.7 mg/l	.5	.001	.05	.0005	.005	.0001
	Silvex[1]	Chinook salmon	LC_{50} 48 hr, 1.2 mg/l	.06	.02	.03	.02	.01*	.01*
	Triazines	Daphnia	LC_{50} 48 hr, 1.0 mg/l	.05	.05	.03	.03	.01	.01
	2,4-D[1]	Bluegill	LC_{50} 48 hr, 1.0 mg/l	.05	.05	.05	.02	.01	.005
	2,4,5-T[1]	Bluegill	LC_{50} 48 hr, 1.4 mg/l	.06	.02	.03	.02	.01	.01

Table 3. (continued)

Class	Chemical	Most Sensitive Test Species Affected	Test Basis & Concentration	Criteria, PPm 24 hr. Mean Stream Class & User					
				< 10 cfs		10 cfs-Navigable		Navigable	
				Potable	Irrig.	Potable	Irrig.	Potable	Irrig.
Herbicide	TCDD	Coho salmon	No effect 96 hr, .000000056 mg/l	--------------.000000006 for all water--------------					
Insecticide	Carbaryl	Stonefly	LC_{50} 48 hr, .0048 mg/l	.001	.001	.0005	.0005	.0002	.0002
	Diazinon	Daphnia	LC_{50} 49 hr, .0009 mg/l	.0001	.0001	.00005	.00005	.00001	.00001
	Disulfoton	Stonefly	LC_{50} 48 hr, .005 mg/l	.001	.001	.00025	.00025	.00024	.00025
	Endosulfan	Rainbow trout	LC_{50} 96 hr, .0003 mg/l	.00003	.00003	.00001	.00001	.000003	.000003
	Endrin	Coho salmon	LC_{50} 96 hr, .0005 mg/l	.00005	.00005	.00001*	.00001*	.000005	.000005
	Fenitrothion	Atlantic salmon	Behavior test 1 mg/l	.025	.025	.01	.01	.005	.005
	Guthion	Stonefly	LC_{50} 96 hr, .0015 mg/l	.0003	.0003	.0002	.0002	.00007	.00007
	Lindane	Brown trout	LC_{50} 48 hr, .002 mg/l	.0001	.0001	.00005	.00005	.00001*	.00001*
	Malathion	Daphnia	LC_{50} 96 hr, .0018 mg/l	.0005	.0005	.0002*	.0002*	.0001	.0001
	Phosphamidon	Daphnia	LC_{50} 48 hr, .0088 mg/l	.0005	.0005	.0005	.0005	.0002	.0002
	Trichlorfon	Stonefly	LC_{50} 96 hr, .016 mg/l	.002	.002	.0005	.0005	.00005	.00005

* As listed in QCW.

[1] The phenoxy herbicides may occur in water as esters or other forms. The given criteria for potable water may be increased by a factor of 10 for forms other than esters. Criteria for irrigation use are for total phenoxy herbicide.

the source is a pentavalent compound. Where natural levels exceed these amounts, recommended maxima are 10 percent increases over background levels for a period not to exceed one year.

DALAPON (2,2-dichloropropionic acid)

Dalapon is a moderately specific grass herbicide used in reforestation weed control. It is water soluble but not notably mobile. *Daphnia* are the most sensitive group of aquatic test organisms showing a response to dalapon (48-hr. LC_{50} 11.0 mg/l). This herbicide is of low general toxicity to mammals, with acute oral LD_{50} on rats estimated at 3860 mg/kg, and LC_{50} 48-hour exposures to fathead minnow, bluegills and coho salmon ranging from 290 mg/l to 340 mg/l. Because of the relative sensitivity of *Daphnia*, the application factor is based on potential damage to aquatic food insects rather than toxicity to potable water users. For maintenance of stream bottom populations, an application factor of 100:1 is used in all but the smallest streams, in which a factor of 20:1 is used in relation to a short term maximum. A margin of safety above that needed for aquatic fauna is required for irrigation water, because dalapon is more harmful to plants, especially grasses, than to animals. This margin may be estimated from records of plant injury and data on plant control.

Dalapon is used as a selective herbicide in crops at rates usually in the vicinity of 51 lbs/acre (55 kg/ha) as the 85 percent active product; in sugar cane, one registered use extends down to 1 lb/acre (1.1 kg/ha) which it is stipulated by the label that only small sensitive weed grasses are affected. A no-effect level to plants cannot be calculated precisely from these data, but it is estimated that a level amounting to one percent of the median of registered use rates, i.e., 0.05 lbs/acre (44 kg/ha), would have a level of activity well below the amounts causing depression of yield. An acre foot of water carrying 0.018 mg/l of dalapon would apply 0.05 pounds per acre (0.055 kg/ha). Dalapon has a biological life expectancy in soil of three weeks, or less, which is shorter than the period over which 12" of irrigation water would normally be applied. Accordingly, the application factor of 100:1 for crop tolerance leads to an allowable level of 0.02 mg/l in sources of irrigation water. Because of the brief nature of concentrations in small streams, higher maxima would not lead to a harmful accumulation on the crop, and a maximum of 0.1 mg/l is recommended.

DICAMBA (3,6-dichloro-o-anisic acid; Banvel)

This chemical is a herbicide of the benzoic acid family. It is of the growth regulator group, and is effective as a brushkiller in some circumstances in which phenoxy herbicides are inadequate. Formulated as amine salts, it is water soluble and somewhat mobile in soil. Dicamba has considerable soil activity, and the compound is relatively persistent. Soil activity can persist for six months at the rate of one pound active per acre (Newton, M., Oregon State University, unpublished observations).

Dicamba is of low acute toxicity to mammals, with acute oral LD_{50} values of over 1000 mg/kg for several species. Effects on fish have not been widely investigated, but the bluegill demonstrates LC_{50} levels of 23 mg/l for 96 hours. Crayfish all tolerate much higher considerations.

Dicamba is used only infrequently for general forestry site preparation. For this reason, it is unlikely to appear in large amounts or in a chronic pattern necessitating a large application factor. Based on bluegills, a 100:1 factor can be used for small streams, permitting a 24-hour mean maximum of 0.2 mg/l. Much less of this persistent growth regulator is tolerable for irrigation.

DINOSEB (4,6-dinitro-o-sec-butylphenol, DNBP, Dinitro)

This compound is a herbicide of the substituted phenol class. It is a desiccant having the effect of contact action on all green vegetation. Its only registered use in forestry is in accordance with state labels permitting its use prior to burning for forest site preparation.

Dinoseb is the most toxic of the herbicides used in forestry. Acute toxicity to mammals is in the very toxic range, of 20-25 mg/kg in mice, rats and guinea pigs and 40 mg/kg for chickens. No data were found in the literature on fish and aquatic insects, but LC_{50} values for 96 hours on scud indicated much lower toxicity to this life form than would be anticipated for an insecticide of comparable toxicity to mammals.

Dinoseb has only recently been registered for use in forestry (Oregon State Label). Despite its long history as an agricultural chemical, data pertaining to aquatic organisms has been difficult to obtain. For the present, the data base is too fragmentary to recommend a specific water tolerance.

PICLORAM (4-amino-3,5,6-trichloropicolinic acid; Tordon)

Picloram is a herbicide of the pyridine family used for controlling brush prior to tree planting. Like dalapon, picloram is a low-toxicity brushkiller in relation to mammals and stream fauna. The most sensitive species reported, largemouth bass, shows an LC_{50} for 48 hours of 19.7 mg/l;

rats, guinea pigs and birds all demonstrate acute oral LD_{50} levels of 3,000 to 10,000 mg/kg. It is relatively persistent and mobile, and a larger-than-average application factor for mammals of 500:1 is proposed. Thus, the criteria for potable water would permit ingestion of $\frac{210}{500}$ g = .42 grams in two liters water, or 210 mg/l. This concentration is known to be severely phytotoxic to trees and broadleaf plants, however, and the allowable concentration suggested reflects a concern for irrigation water rather than a problem with acute toxicity to animals. Quality criteria will therefore be based entirely on likelihood of injuring aquatic and crop plants.

Picloram is among the most active of the herbicides. Dosages of one-half gallon per acre (4.7 l/ha) of Tordon 101 herbicide (containing .263 kg/ha picloram) are recommended as a minimum for certain weed problems. One pound picloram per acre (1.15 kg/ha) is approximately the median dose, and it is usually mixed with four pounds 2,4-D. Applying an application factor of 100:1, or 0.01 pounds per acre (0.011 kg/ha), would permit a concentration of 0.004 mg/l for one foot of irrigation water. Bruns et al. (1972) observed that 0.040 mg/l in irrigation water caused slight injury to cotton in three subsequent irrigations, but did not cause a reduction in yield. Legumes, tobacco and potatoes are more sensitive than cotton. The latter, especially, are extremely sensitive. Dr. Alex Ogg, U.S.D.A., Prosser, Washington, (personal communication, 1976) has observed damage typified by malformed stalks and stunting, as well as malformed growth from tubers thus treated, when potatoes were irrigated for an extended period with water containing picloram at .0005 mg/l. It appears that for picloram and other growth regulating herbicides, an application factor of 100:1 below median dosage is inadequate. The adoption of .0001 mg/l as a criterion for irrigation water over long periods poses analytical difficulties in monitoring. It is necessary, however, to insure some measure of safety for any irrigator. The level of .0001 mg/l is tentatively proposed for irrigation water over extended periods. Concentration maxima of .001 mg/l proposed for short term peaks in feeder streams would be unlikely to provide a quantity of contaminated irrigation water that would be likely to cause injury, although some malformation could occur without loss of yield. Potato bioassay may be the most sensitive analytical technique for monitoring.

SILVEX (2-(2,4,5-trichlorophenoxy)propionic acid)

Silvex is a phenoxy herbicide used for several specific brush control jobs in forest site preparation and release. Like the other phenoxys, it is neither persistent nor mobile. This material has been registered for many years as an aquatic herbicide. Its effects have been tested on a wide range of aquatic and warm blooded species. Silvex is of medium to low toxicity among mammals and birds, with acute oral LD_{50} values in the range of 600 to 2,000 mg/kg. The range of concentrations at which fish have demonstrated LC_{50} concentrations for 48 hours extends from 1.23 mg/l for the chinook salmon to 83 mg/l for bluegills. Formulation is important, with esters tending to be more toxic to fish than salts. Because esters are more widely used in forestry than are other formulations, and fish appear to be among the more sensitive organisms, the safety factor is based on salmon responses to esters, rather than on the expectation of mammals being affected by drinking water. Of the esters, those containing butyl groups in the alcohol chains appear to be most toxic to fish. These include the butyl, butoxy ethanol, propylene glycol butyl ether esters and perhaps others.

A long-term application factor of 100:1 is proposed to allow for the most sensitive fish being exposed to a butyl-containing ester; i.e., 0.01 mg/l is proposed for the maximum concentration in large streams; this corresponds to the levels recommended by the QCW, and is one-third of that suggested by Rohlich (1972) for potable water. Higher maxima are recommended for smaller streams on the principal that exposures are of very short term. Moreover, the extensive experience with silvex as an aquatic herbicide has not led to obvious problems, suggesting that short term exposure to one-hundredth the registered aquatic weed control dosage will not lead to long term injury.

Silvex in irrigation water has been studied by Bruns et al. (1973), who reported that concentrations in the range suggested for potable water do not cause yield reductions for sugarbeets, soybeans, or corn. Soybeans, the most sensitive crop studied, were adversely affected at 0.02 mg/l in irrigation water, in which the principal effect was a slight reduction of seed quality, without reduction in yield. No effects were observed below this level. Growth regulator activity is apparently present to a lesser degree than in 2,4-D and picloram.

Silvex is one of the several products made from intermediates containing TCDD. This compound will be considered separately in relation to its own toxicity and criteria. Its concentration in silvex is regulated by mutual consent among manufacturers at 0.1 mg/kg of silvex acid, at which concentration its contribution to toxic hazard is well below that of silvex, despite its extreme absolute toxicity. It will therefore not be considered in the determination of silvex criteria.

S-TRIAZINES (atrazine; simazine)

Atrazine and simazine are low-solubility, low mobility herbicides whose biological effects are generally expressed in the inhibition of photosynthesis. They are widely used for herbaceous weed control in plantations. They are of generally low toxicity to warm blooded animals and fish, with warm blooded animal acute oral LD_{50}'s in the range of 1,750 mg/kg for atrazine fed to rats and mice, to 5,000 mg/kg, also for rats, exposed to simazine. Long-term cattle-feeding studies with 750 mg/kg/day of simazine caused weight loss with no mortality. Simazine is registered as an aquatic herbicide.

Fish are quite tolerant of the triazines. Tests with various game-fish have placed 48-hour LC_{50}'s in the range of 4.5 mg/l for rainbow trout to 118 mg/l for bluegills. Daphnia, however, were affected by lower concentrations of simazine, with 1 mg/l for 48 hours an estimated LC_{50} (Sanders, 1970). In terms of potential harm to fauna, then, it is most probable that the triazines will exert their effects primarily at the lower trophic levels. Because of the resiliency of this group of organisms and the general tolerance of most of the vertebrates, the application factor of 100:1 could be based on the Daphnia, allowing large river concentrations of 0.01 mg/l. Much higher concentrations (5.0 mg/l) are used in aquatic weed control, and the above criteria should not be construed as limiting use of the triazines for that purpose.

The generally specific nature of triazines as herbicides suggests that they may require more stringent limitations in irrigation water than in water used for other purposes. The triazines are not considered growth regulator herbicides, and a safety factor can logically be derived from recommended use rates. Simazine and atrazine are generally recommended for use between 2 and 4 lbs/acre (2.2 to 4.4 kg/ha), with a median dosage of about 2 1/2 pounds, active, (3 kg/ha). Using the same factors and calculations as for dalapon, a safe concentration level is calculated at 0.01 mg/l, the same as for potable use, and equal to the level recommended for aquatic systems by Rohlich (1972).

Triazines are more persistent than dalapon, and the use of calculations based on the shorter residual compound should be examined in terms of available data on crop sensitivity. Ries _et al_. (1967) studied the responses of several crop species to triazines at application rates in the range of grams per acre. He observed that crop yields were not substantially affected by these low rates, but that the protein contents were increased. His studies were conducted on several species of grains as well as on peas, all of which appeared to tolerate the proposed concentration without harm. The calculated long-term maximum of 0.01 mg/l is therefore recommended for irrigation water. The higher levels proposed for smaller streams reflect the likelihood of transience of contamination. The upper limit of 0.05 mg/l in feeder streams is substantially below the LC_{50} for the most sensitive aquatic species known, and provides an application factor of 10,000:1 or more for potable water users.

2,4-D (2,4-dichlorophenoxyacetic acid)

This phenoxy herbicide is one of the most commonly used herbicides in forest weed and brush control, aquatic weed control and in the growing of grain crops, and has been extensively tested for toxicity on many organisms. It is immobile in soil, and is degraded rapidly in plants and soil. It is in the medium toxicity range, with acute oral LD_{50}'s for warm blooded animals in the range of 368 mg/kg for mice to 2,000 mg/kg for mallards. Toxicity to fish is moderate to low, depending on formulation. The acid of 2,4-D is very low in toxicity, with LC_{50} values ranging from 375 mg/l for bass to 390 mg/l for bluegill in 48-hour tests. Formulation as esters, however, decreased the LC_{50} values for bluegills to a range of less than 1.0 to 9.0 mg/l, depending on which ester was used; the low volatile esters most commonly used in forestry ranged from 3 mg/l to 9 mg/l. As with silvex, esters containing the butyl group appear to be the most toxic. Stoneflies, with LC_{50} values for the butoxyethanol ester of 1.6 mg/l, were comparable in sensitivity to bluegills; crayfish were very low in sensitivity, with 48-hour exposure to 100 mg/l producing no visible effects. Meehan et al. (1974) reported extreme sensitivity of Alaska salmon species to the butyl ester, which is not registered for forestry use.

Fish are apparently the most sensitive group of animals in response to direct exposure. This, plus the extensive use of 2,4-D, suggest the use of a moderately large safety factor when using ester formulations. Based on LC_{50} value of 1 mg/l and a 100:1 application factor a sustained river concentration of .01 mg/l would be "safe" for fish and furnish a very large margin of safety for potable use. For 2,4-D amine, .1 mg/l would be permissible with a larger safety factor, and it is recommended that tolerance levels of 2,4-D acid or amine be higher than for the esters by a factor of ten except for irrigation uses. Because hydrolysis of the esters to acids begins to occur immediately, it is recommended that the efforts to monitor make the distinction between forms in each analysis. The stated criteria for potable use apply only to the ester fraction.

Irrigation water must consider the growth regulating properties of 2,4-D. Beans, potatoes, grapes and cotton are severely affected by very low rates of 2,4-D. Unlike picloram, however, 2,4-D is rapidly destroyed by metabolic activity in the plants, and it is not as damaging to sensitive crops at extremely low concentrations as is picloram. Studies of 2,4-D in irrigation water have been conducted with sensitive crops. Concentrations of 0.22 mg/l caused visible symptoms but no yield reductions in soybeans and sugar beets. Concentrations of 0.022 mg/l did not injure any crop visibly. Crops did not carry detectable residues (Bruns et al., 1973).

Because of the growth regulator properties of 2,4-D, concentrations in irrigation water have to be maintained at lower levels than would be dictated for drinking water or fish habitat. The proposed level of 0.005 mg/l provides for a margin of safety of 5:1 below the apparent minimum limits of crop sensitivity; the increased concentrations permitted in feeder streams will apparently not exceed levels at which aquatic communities will be damaged, and they are lower than the suggested maxima listed in the QCW, or recommended by Rohlich (1972).

2,4,5-T (2,4,5-trichlorophenoxy acetic acid)

2,4,5-T is the phenoxy most widely applied by aircraft for brush control in the reforestation of forest lands. The toxic hazards associated with 2,4,5-T have recently received intensive study. Because of its wide use in forest management, it is among the most likely to find its way into forest waters through direct application to water surfaces.

This compound is very closely related to silvex, both chemically and toxicologically. It is non-persistent in soil (14-30-day half life) and low in mobility (Norris, 1970; Blackman et al., 1973). It is used at similar rates of application, has a similar spectrum of biological activity and is also applied in identical formulations.

Teratological studies have dominated the adverse findings regarding 2,4,5-T. Courtney et al. (1969) reported that 2,4,5-T caused birth defects in mice receiving dosages of 46-113 mg/kg/day for 9-10 days during early pregnancy. Their findings were tempered by the discovery that their 2,4,5-T contained 27 parts per million TCDD (Leng, 1974), but later reports have confirmed that levels of 2,4,5-T high enough to cause distress to the dams can cause some anomalies in the litters even with TCDD levels at 0.1 mg per kg 2,4,5-T acid or less, the prevailing purity standard (Courtney et al., 1970). Goldberg (1970) does not regard their data as evidence that low-TCDD 2,4,5-T is a teratogen. Because of the high levels used in these toxicity tests, and the evidence of toxic effect at these dosages on the mothers, the question has not been resolved as to whether 2,4,5-T is a teratogen, or whether fetal distress was the result of maternal stress. In any event, embryotoxic effects have not been demonstrated with single dosage or at chronic dosages substantially below those producing direct toxicity symptoms.

The dosages at which harmful effects have been reported place 2,4,5-T in the medium-toxicity class. Rowe and Hymas (1954) reported acute oral LD_{50} levels for rats at 495-750 mg/kg, similar to silvex. Guinea pigs are apparently more sensitive to 2,4,5-T than to silvex, with acute LD_{50} levels of 380 mg/kg for 2,4,5-T as compared to 1,250 mg/kg for guinea pigs. Dogs are reportedly among the most sensitive animals in acute oral dosage tests, with LD_{50} levels in the vicinity of 100 mg/kg.

Feeding studies have shown mammals to pass 2,4,5-T through their digestive systems without substantial uptake or modification. Cattle and sheep were fed at the rate of 100 mg/kg/day for cattle and 50 mg/kg/day for sheep for ten days without observable effects. Chronic levels needed to cause minor weight loss in dogs was 10 mg/kg/day. Because much of toxicity data for 2,4,5-T were reported before the discovery of the counfounding from TCDD and the reduction of the concentration of TCDD in production grades of herbicide, it is likely that the values reported provide some additional margin of safety.

The effects of 2,4,5-T formulations on fish are very similar to those of silvex. LC_{50}'s of 1.4 mg/l are reported for the most toxic esters on bluegills. Again, there appears to be a wide range of toxicity among the esters, and amine formulations are substantially less toxic than any ester; standards for the forms other than esters are suggested at ten times those of the esters. For this reason, monitoring of 2,4,5-T should identify esters distinct from other forms. Because the patterns of 2,4,5-T effects on various species are similar to silvex, the same rationale is used for recommending water quality criteria, and the same levels are proposed. These range from .01 mg/l in large streams to .06 mg/l in feeder streams. These concentrations are higher than those listed by Rohlich (1972) on the basis that concentrations of TCDD in production grade 2,4,5-T are considerably less than one tenth of those assumed to be present in his calculations. Moreover, the question of TCDD is handled separately here, and either 2,4,5-T or TCDD criteria may be used to determine the need for pollution control.

TCDD (2,3,7,8-tetrachlorodibenzo-p-dioxin)

TCDD is an unavoidable contaminant in all pesticide products in which 2,4,5-trichlorophenol is used as a manufacturing intermediate. It is a highly insoluble and immobile material that is persistent in soil but rapidly degraded in vegetation (Crosby and Wong, 1976). This material is among the most toxic synthetic substances known to man. The amounts which are present in 2,4,5-T and silvex sold as commercial products is controlled by the manufacturers to have concentrations of 0.1 mg of TCDD per kilogram of phenoxy acid equivalent, or less. Unpublished data from the EPA Dioxin Implementation Program indicated that current 2,4,5-T production samples were in the vicinity of .01 mg TCDD per kg 2,4,5-T, or less.

TCDD apparently does not figure substantially in the acute oral mammalian toxicity of phenoxy herbicides meeting the 0.1 ppm TCDD concentration standard. Rats, mice and rabbits demonstrate acute oral LD_{50}'s of 0.022 mg/kg, 0.114 mg/kg and 0.115 mg/kg respectively. Guinea pigs are extremely sensitive, with an acute oral LD_{50} of 0.0006 mg/kg. The ratio of toxicity of TCDD to 2,4,5-T is greater for guinea pigs than for other mammals studied. at 1:750,000; for many species the ratio is about 1:100,000. Since the ratio of occurrence of 2,4,5-T to TCDD is 10,000,000:1 or greater, there is little likelihood of being able to observe TCDD effects when administered in a phenoxy vehicle.

Fish are apparently the most sensitive organisms among those studied for the effects of TCDD. Miller (1974) did, however, discover an apparent level at which coho salmon in glass environments tolerated TCDD without measurable effects on weight gain or other vital functions. This dosage was

.000056 mg/kg, based on mg TCDD in an aquarium per kilogram live weight of fish. Over a 60 day period, however, Miller et al. (1973) had observed an LC_{50} for rainbow trout of 0.056 ug/l in solution, and an absence of chronic effects in fish diets. Miller's work was remarkable in demonstrating that a 4-day exposure at these low levels produced mortality that continued for 60 days. These data suggest that fish can accumulate amounts of TCDD that produce symptoms after withdrawl but before they can be eliminated.

Isensee and Jones (1975) investigated the tendency for this chlorinated hydrocarbon to be magnified by biological systems. Their data led to the conclusion that organisms exposed to an environment containing a given concentration of TCDD would develop concentrations of TCDD substantially higher than that of the water. Miller's findings (1974) indicate that fish in a glass-enclosed system may function as a sink for TCDD, but consumers at higher trophic levels were not found by Isensee to accumulate substantially increased levels as the result of ingesting prey with elevated TCDD levels in or on their bodies. The tendency for TCDD to accumulate in mammals is similar to that of other chlorinated hydrocarbons. Rose et al. (1976) demonstrated that chronic feeding of rats with .00001 to .001 mg/kg/day led to equilibrium concentrations in 13 weeks. After withdrawal, residues in the rats declined with half lives of 23.7 days, regardless of dosage. This strong evidence of accumulation is thus coupled with a tendency for elimination or detoxication systems to continue functioning despite prolonged exposure. Following the study by Norris and Miller (1973) indicating lethality to guppies of .0001 mg/l in water, Norris[1] has indicated in personal communication that an estimated no-effect concentration of TCDD in water inhabited by guppies or coho salmon is between 0.05 and 0.005 parts per trillion, (.00000005-.000000005 mg/l).

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[1] Norris, L. A. U.S. Forest Service, Forestry Sciences Laboratory, Corvallis, Oregon.

The toxicity of TCDD in aquatic systems is very difficult to evaluate, and safety estimates must remain at least partly conjectural. This substance is extremely insoluble in water, at 0.0002 mg/l solubility. It has a very strong affinity for some surfaces (Miller, 1974). Stream-bottom rocks, plants and organic debris probably bind TCDD so extensively that it is difficult to maintain a solution of TCDD at a steady state for investigation. For the very same reason that such investigations are nearly impossible in other than clean aquarium systems, the material appears unlikely to remain in stream water in detectable concentrations. The rationale in proposing a maximum concentration of .006 part per trillion (.000000006 mg/l) for all water is based on the existence of data suggesting that this level is unlikely to cause effects, and the evidence that a concentration of this order would be unlikely to persist more than briefly. It is also far below the limit of detection. Quality control can, however, be achieved by observing contamination limits for 2,4,5-T and silvex containing 0.01 mg TCDD/kg acid equivalent or lower levels accordingly for more contaminated herbicide, provided the calculated amount is not exceeded.

CARBARYL (1-naphthyl N-methylcarbamate; Sevin)

This insecticide is of the carbamate group, and has found wide usage for control of forest defoliators because of safety, short residue, and restrictions on use of chlorinated hydrocarbons. It is of medium mammalian toxicity, with acute oral LD_{50}'s ranging from 280 mg/kg in the guinea pig to

710 mg/kg in the rabbit. Carbaryl is lower in toxicity to fish than are many of the insecticides used in forestry. Macek and McAllister (1970) report 96-hour LC_{50}'s for eight species of fish ranging from 0.764 to 20.0 mg/l; coho salmon were the most sensitive while catfish were the most resistant.

Arthropods are quite sensitive to carbaryl. Stonefly LC_{50} is 0.0048 mg/l and crayfish only slightly less sensitive at 0.0086 mg/l. Dungeness crabs are less sensitive, having an LC_{50} for 96 hours of 0.180 mg/l. Other estuarine organisms, including two species of oyster and bay mussel tolerate carbaryl at much higher concentrations, with LC_{50}'s of 2.2 and 3.0 mg/l.

The sensitivity of species on which fish depend for food suggests that criteria be directed toward the protection of this portion of the aquatic system. Because carbaryl is a non-accumulating insecticide of short residual life, the criteria can be set so as to prevent a pulse concentration from causing short term damage to food arthropods. An application factor of 20:1 will prevent mortality of food insects. If the application factor of 20:1 is to be used in large rivers, the resulting maximum concentration of .0002 mg/l would give a factor of 900:1 for Dungeness crab and more than 2,500:1 for other members of the aquatic ecosystem covered in this review. Somewhat higher concentrations could be tolerated in feeder streams with minimal effect on benthic organisms. Concentrations of carbaryl at these levels would not influence irrigation water quality. These concentrations are higher than those recommended by Rohlich (1972) on the basis that chronic exposure is unlikely.

DIAZINON (0,0-diethyl 0-(2-isopropyl-6-methyl-4-pyrimidinyl)phosphorothioate)

This insecticide is of the organophosphate class. It is used very little in forest crops, and does not have general registration for aerial application. It is considered here because of its occasional use on ornamentals, especially Christmas trees, near inhabited areas.

Diazinon is regarded as moderately toxic to laboratory animals via oral, dermal and inhalation routes. Chronic effects were not observed in rats fed diets containing 100 ppm for two years, nor in dogs fed 5.2 mg/kg/day for 43 weeks. Monkeys fed at rates of 5.0 mg/kg/day for 106 weeks showed cholinesterase inhibition but no other symptoms. Tolerances in food have been set in most food and vegetables at 0.75 ppm (Von Rumker _et al_., 1974). There appears little likelihood that harm to humans will result from registered usage.

Certain aquatic organisms and birds appear to be more sensitive to diazinon than are mammals. Rainbow trout and bluegills apparently can tolerate high levels, with LC_{50} levels ranging from 30 to 170. mg/l for 48 hours. Stoneflies are slightly more sensitive, with LC_{50} values of 25 mg/l for 96 hours. The oral acute LD_{50} to birds, including the mallard, ranged from 2-4.3 mg/kg. Sanders and Cope also reported that _Daphnia_ LC_{50} was 0.9 mg/l for 48 hours.

Von Rumker _et al_. (1974) observed that diazinon is moderately persistent in the environment. Because of the tendency to stay in place, they concluded that this material is unlikely to cause harmful effects in aquatic systems away from the target area. The finding that the most sensitive group of species, _Daphnia_, shows no effect from exposure to 0.26 mg/l lends credence to this general conclusion.

Using an application factor of 100:1, (because of persistence) from the concentration causing 50 percent mortality in the most sensitive species with a two-day exposure, a large-river concentration of .00001 mg/l would appear to provide a substantial margin of safety for fish and food species. Because irrigation water containing diazinon is unlikely to cause damage or illegal residues in crops, the same standard can be used for potable and irrigation uses. This concentration is close to that suggested by Rohlich (1972), but higher maxima are proposed for smaller streams with less likelihood of chronic exposure.

DISULFOTON (0,0-diethyl-5-(2-(ethylthio)ethyl)phosphorodithioate)

This insecticide is of the organophosphate group, and is considered in the very toxic to extremely toxic range. The material is registered only for use on ornamentals, including some forest species, and has not been used on a massive scale.

Disulfoton has been studied for its toxicity to rats, birds, fish and aquatic insects, among other organisms. There is a large amount of variation in sensitivity among groups of animals. Acute oral LD_{50} for rats is 12.5 mg/kg. Bobwhite quail are far less sensitive, with LD_{50} at 800 mg/kg, but fish can be quite sensitive. Pickering (1962) reported that fathead minnows are very sensitive to disulfoton, with 96-hour LC_{50} levels at 0.063 mg/l. He observed, however, that bluegills were considerably more resistant, with the LC_{50} level at 3.7 mg/l. Stoneflies are highly sensitive; the LC_{50} level for 48 hours is 0.005 mg/l and the LC_{50} for 30 days is apparently about one third the level expected for 24 hours.

Recommended criteria for disulfoton are based on the rationale that this compound is neither persistent nor mobile, and its use pattern virtually prevents the occurrence of any chronic exposure. The maximum concentration in large rivers is suggested at .00025 mg/1, which is one-twentieth the concentration known to cause mortality in a two-day exposure to the most sensitive group of organisms. Because of the non-accumulation characteristics of this compound, and because this concentration allows for a 250:1 application factor for a sensitive fish species, a short term exposure at this concentration should be completely compatible with freshwater fisheries and ecosystem productivity. The only departure from this level is suggested for feeder streams, in which an allowable short-term maximum of 0.001 mg/1 is proposed. The upward deviation from Rohlich's (1972) proposed maxima is based on the exclusively pulse-type concentrations likely to occur.

ENDOSULFAN (6,7,8,9,10,10-hexachloro-1,5,5a,6,9,9a-hydro-6-methano-2,4,3-benzo(e)-dioxathiepin-3-oxide; Thiodan)

This organochlorine insecticide is only used on ornamentals and on specialty forest crops, such as Christmas trees. It is moderately persistent, and is used for control of needle miner insects and others for which extended effectiveness is needed. Its use is so limited that its consideration here is a reflection of the concentration of those uses near human habitation, and its potential for harm to fish if deposited in water.

Of all the insecticides registered for use on forest species, endosulfan may have the greatest potential for causing injury to fish from any given application. Its use in Christmas tree plantations is superimposed on a weed control system in which there is maximum opportunity for surface runoff, which is not normally an important route of stream contamination in forests. It is moderately persistent, but has very low solubility. Therefore, the principal route of entry into streams is likely to be through sediment transport, in which form most of the material is not in solution.

Rainbow trout are extremely sensitive to endosulfan. The LC_{50} of 0.0003 mg/l for rainbows indicates that the freshwater fishery is very vulnerable to contamination by this substance. Because of the potential persistence of the chemical, low chronic levels are a possibility. In order to minimize the likelihood of chronic exposure, the maximum concentration of dissolved endosulfan in rivers is suggested at .000003 mg/l, the same as proposed by Rohlich (1972), providing a concentration maximum with a 100:1 application factor. Because of the low solubility of endosulfan and its infrequent use, it is unlikely that concentrations would build up in food organisms if no pulse concentrations exceed this level.

Maximum concentrations in feeder streams are suggested at 0.00003 mg/l. This maximum allows for short term peaks that are considerably below the lethal level for fish. No greater allowance is suggested for safety because the other aquatic organisms are largely more tolerant of the insecticide than are the fish that feed on them.

Monitoring of solution levels will require filtration or centrifugation to remove adsorbed material prior to extraction.

ENDRIN (Hexachloroepoxyoctahydro-endo, endodimethanonaphthalene)

This chlorinated hydrocarbon insecticide has been used in forestry primarily to protect directly sown conifer seeds from seed-eating rodents. Endrin is low in water solubility and mobility. Quantities of endrin applied in this manner range from 2.5 to 10 gms per hectare, much less than for most forest insecticides.

Endrin is extremely toxic to fish and fish food organisms, with 96 hr LC_{50}'s reported in the .0001 mg/1 to .003 mg/1 range. Mammals are somewhat less sensitive (1-16 mg/kg). Criteria will therefore be recommended to protect aquatic communities.

Maximum concentration recommended for the smallest stream category is .00005 mg/1. For intermediate streams, a much lower concentration of .00001 mg/1 is recommended due to endrin's potential for bioconcentration (Mount and Putnicki, 1966). In large rivers, a .00005 mg/1 limit is proposed, for the same reason, providing a 100:1 application factor for coho salmon. Because endrin is eliminated from fish soon after exposure ceases (Argyle et al., 1973) these limits should protect against disruption of aquatic communities. The large river criteria suggested here is slightly higher than that recommended by Rohlich (1972). Centrifugation or filtration should be a part of the procedure for water monitoring.

FENITROTHION (0,0-dimethyl 0-(4-nitro-m-tolyl)phosphorothioate; Sumithion)

This insecticide is of the organophosphate group, closely related to parathion in structure. In contrast to parathion, fenitrothion is only moderately toxic to mammals, and the compound has been extensively used as a substitute for DDT and parathion for this reason. Fenitrothion has been applied to several hundred thousand acres under temporary registration to control spruce budworm (Gooley, 1973) and is now fully registered. Many million acres in eastern Canada have been treated with fenitrothion for the same purpose.

Feintrothion selectivity in forests is based on the generally low toxicity to mammals, with acute oral LD_{50} in rats being 250-670 mg/kg. Also, fenitrothion was not generally found to be metabolized to fenitroxon, although it depends for its insecticidal effects on the ability of insects to make such a transformation (Miyamoto et al., as cited by Matsumura, 1975).

Toxicity to salmonid fish is sufficiently low that research has been concerned with behavioral manifestations. Symons (1973) reported that 50% of the young salmon exposed to 1 mg/l for 16 hours ceased to maintain territories. Wildish and Lister (1973) found a loss of hierarchical order among brook trout which were fed a diet containing 10 mg/g of fenitrothion for 4 weeks. When treatment ceased, fish behavior returned to normal in both cases.

Since toxicity to fish is relatively low, the most likely effect of this material would be to aquatic insect populations, for which toxicity data are scant. Tracy and Purvis (1976) monitored the effects on aquatic insects of applying 2 to 4 oz./acre (.14 to .28 kg/ha) of fenitrothion to portions of six forested watersheds in north central Washington. Buffer strips along streams were not excluded. Peak fenitrothion concentrations ranged from 1 to 30 ug/liter. At the highest measured concentration, 20% mortality of mayflies occurred, but the only change noted at most sampling sites was a temporary increase in the proportion of mayflies among drifting live insects. An application factor of 200:1 is therefore recommended below the lowest response level by fish. The recommended maximum in large streams is therefore .005 mg/l.

GUTHION (0,0-dimethyl S-(4-oxo-1,2,3-benzotriazin-3(4H)-ylmethyl)phosphorodithioate); (azinphosmethyl)

This insecticide is of the organophosphate group with some residual action and ability to penetrate skin. It is very toxic to rats and aquatic organisms. Its limited use in forestry prevents substantial exposure of any forest wildlife, but it may be used on Christmas trees and other ornamentals where opportunity for human exposure is greater. Considerable difficulty will be encountered in identifying pollution specifically related to silvicultural uses, because it is widely used on orchard crops and seldom on forest species.

Guthion limitations in water can be based on expectation of harm to fish and aquatic insects. Among the most sensitive fish are salmon. 48-hour LC_{50}'s for coho and chinook salmon are 0.005 and 0.0062 mg/l. Stoneflies are more sensitive, with 96-hour LC_{50} of 0.0015 mg/l. Differences in exposure time have little effect on response to a given concentration suggesting that the critical exposure period is brief. Criteria for small streams should therefore be close to those of rivers. Numerous fish species, especially those native to warm water, are apparently less sensitive than salmon, and the rationale for setting criteria is based on food organisms. On this basis, a maximum concentration in large rivers of 0.0007 mg/l is proposed. This concentration provides a 20:1 application factor for the most sensitive known fish. Slightly higher limits are proposed for large streams capable of supporting a freshwater fishery, and for feeder streams. The character of this insecticide suggests that these concentrations would cause little insect mortality, and would be of short duration. Rohlich (1972) recommended maxima of .000001 mg/l, much lower than is proposed here.

His rationale is not clear for the extremely low level unless there should be evidence that groups of aquatic insects are substantially more sensitive than the stonefly. In this event, the above recommendation should be altered accordingly.

LINDANE (Gamma isomer of 1,2,3,4,5,6-hexachlorocychohexane)

This organochlorine insecticide is one of the few in its group that has appreciable solubility in water (10 mg/l). Its use in forests is limited to application as a bark spray for control of bark beetles. The absence of aerial applications virtually precludes its appearance in streams despite its widespread use for control of the southern pine beetle.

Lindane is moderately toxic to mammals, with acute oral LD_{50}'s in the range of 60-127 mg/kg. Due to rapid metabolic breakdown, it has low potential for bioaccumulation (Koerber, 1976). The toxicity of lindane has been investigated on a large number of fish and aquatic organisms. LC_{50}'s range from 0.002 to 0.083 mg/l for nine species of fish, with brown trout being the most sensitive by a substantial margin. Stoneflies are also sensitive, with LC_{50} of 0.0045 mg/l, 100 times more sensitive than Daphnia. On the basis that a prime freshwater fish is the most sensitive known non-target species, it will be used to determine the criterion for limiting concentration.

In proposing 0.00001 mg/l as the maximum concentration in rivers, a direct application factor of 200:1 is used. The very large factor is based on the potential for chronic exposure at extremely low levels resulting from the persistence of this compound and solubility properties that render it capable of moving slowly into streams. Higher concentrations are proposed for large streams and feeder streams on the basis that they still provide a margin of 20:1 or more, and their rates of flushing do not permit a sustained elevated level. This level is the same as proposed in the QCW.

Although the use of lindane as a general insecticide is decreasing, its use in other than forest operations far outshadows the very small amount used for beetle control. Under the circumstances, the total contribution from forest operations is subject to masking by other much larger sources. There is therefore concern in setting criteria for forest use that monitoring and enforcement pose serious practical problems in relation to the magnitude of the potential danger from contamination by other uses. Therefore, water sampling should be done both above and below projects along a stream, if possible. Samples should be filtered or centrifuged to remove particles.

MALATHION (0,0-dimethyl phosphorodithioate of diethylmercaptosuccinate)

This insecticide of the organophosphate group is well known for its safety and selectivity among terrestrial species. Because of its record of safety, lack of persistence and utility as a general insecticide, malathion may eventually find more general usage in forests, where it is used little at present.

Matsumura (1975) describes malathion selectivity for mammals in terms of the ability of the mammalian liver to break down the compound by removal of its carboxyl group. Because of this ability, the acute oral LD_{50} in rats has been reported at the high levels of 597 mg/kg to 5,800 mg/kg. Fish are less sensitive to malathion than to many other insecticides, but are nevertheless quite sensitive. Ninety-six hour LC_{50}'s for fish among the salmonids range from 0.1 mg/l to 0.2 mg/l. Other fish in the same range include bluegill, redear sunfish, largemouth bass, and perch, while fathead minnow and channel catfish show the surprisingly high 48-hour LC_{50} levels of 8.97 and 9.0 mg/l respectively.

Long-term effects of malathion on fish have been studied both in terms of chronic toxicity and determination of no-effect levels. Some chronic (10 month) effects were observed in fathead minnows and bluegills at concentrations approximately 1/15 of the 96-hour LC_{50}'s. Concentrations of 1/30 to 1/45 of the 96-hour LC_{50} produced no responses during exposures of 10-11 months. No-effect levels for the most sensitive fish have been identified at 0.0036 mg/l.

The stonefly is apparently more sensitive than is any fish. The LC_{50} level (96-hour) for stoneflies is 0.01 mg/l. Daphnia is more sensitive, with a 96-hour LC_{50} of 0.0018 mg/l.

The rationale for establishing a maximum concentration of malathion in water is based on potential injury to aquatic food organisms. Because malathion is non-persistent and non-accumulating, and because aquatic insects have substantial ability to recover after some damage to their populations, a smaller safety factor can be used than would be needed for a more persistent compound. Moreover, the potential harm from a brief spike in a feeder stream is small enough to permit the occurrence of such events at levels approaching one fourth of the 96-hour LC_{50} levels without causing harm to the freshwater fishery. Under the circumstances, application factors of 20:1 for Daphnia and 100:1 for stoneflies are proposed for establishing maximum levels in rivers, leading to a recommended maximum of 0.0001 mg/l, which is below the maximum suggested in the QCW. This maximum could be raised in streams of feeder classes to 0.0005 mg/l without increasing the likelihood of injury to the fishery, provided these exposures are limited to 24 hours, and without risking dangerous levels in major rivers.

PHOSPHAMIDON (2-chloro-2-diethylcarbamoyl-l-methyl vinyl dimethylphosphate; Dimecron)

This systemic insecticide is one of the organophosphates used for aerial control of forest defoliators. It is water soluble and non-persistent. Although very toxic to mammals it is even less toxic to some fish than carbaryl. Sensitivity of insects and crustaceans is variable, but some species are much more sensitive than fish. Daphnia (water fleas) were the most sensitive organisms tested (96 hr LC_{50} of .0088 mg/l) while crayfish were more resistant than some fish species (96 hr LC_{50} of 7.5 mg/l).

Phosphamidon is not widely used in forestry at present but is being investigated as a substitute for DDT against defoliating insects. It may eventually be applied over large areas, if found to be effective.

Criteria will be set to protect food species, since some of them are orders of magnitude more sensitive than fish. A concentration of .0005 mg/l in feeder streams would provide a 96 hr application factor of about 20:1 for the most sensitive species tested and 40:1 for the second most sensitive species. Since this chemical is neither persistent nor cumulative, limits for larger streams do not need to be much more stringent. A concentration of .0002 mg/l provides nearly a 50:1 application factor for the most sensitive species tested. This level is somewhat higher than that suggested by Rohlich (1972) and is given on the basis of the excellent safety record of this insecticide.

TRICHLORFON (Dimethyl(2,2,3-trichloro-1-hydroxyethyl)phosphonate; Dipterex)

This insecticide is one of the organophosphate group used for aerial control of defoliators, and is used agriculturally in animal feed (as a bloodstream systemic) as well as to protect plants. It is effective against insects because it converts to dichlorvos within the insect's digestive tract. It is moderately toxic to mammals (LD_{50}, 400 and 600 mg/kg for rat and mouse, respectively) and is rapidly excreted in urine. Toxicity to insects and crustaceans is 10-1000 times greater than for fish, therefore criteria will be recommended so as to protect fish food organisms rather than the fish. Toxicology of trichlorphon itself is similar to phosphamidon and the same concentration for feeder streams is recommended. Because trichlorphon is converted to dichlorvos (which is much more toxic to crustaceans and somewhat more so to fish) under mildly alkaline conditions (pH 7 and above) much more stringent limits have been set for larger streams. A maximum concentration of .00005 mg/l for large streams provides a small safety margin for the most sensitive crustacean tested and a 5 to 10 fold margin for the majority, assuming complete conversion to dichlorvos. Since most forest streams are acid, this margin is considered conservative.

Trichlorphon has been tested against gypsy moth and spruce budworm on an operational scale. A possibility therefore exists that it may be applied to very large areas if these test results favor it over other insecticides. The recommended concentration limits by Rohlich (1972) proposed a maximum of .000002 mg/l. The presumption is made that such stringent limits are proposed to protect against alkaline conversion. On this basis, the .000002 mg/l limit is supported for streams having pH of 7.0 or greater.

BEHAVIOR OF CHEMICALS USED IN SILVICULTURAL OPERATIONS

Toxic hazard and other adverse effects of chemical pollution have been described principally in terms of concentration in water, and duration of elevated levels. Maintenance of water quality will depend on use patterns which result in minimum entry of chemical into streamflow, and minimum harmful effects from any chemical that inadvertently reaches the water.

Operational guidelines for chemicals have several important points at which control is exerted over the probability that harmful effects to aquatic systems and users will occur. These are 1) control over the proximity of application to streams, 2) control of dispersion from application equipment, and 3) control over choice of chemical, hence mobility and ability for a given application rate to cause damage. The effectiveness with which these controls are prescribed depends on characteristics of the equipment, chemical, and the environment into which it is introduced. Specific control guidelines must be grounded on scientific evaluation of interaction among physical, chemical and biological systems combined with observational experience gained from monitoring chemicals and affected biota. The purpose of this chapter is to review the behavior of the applied chemicals, and to provide guides as to where and how the chemicals can be applied without affecting water quality.

Physical Properties of Silvicultural Chemicals in Relation to Mobility in Soil

FERTILIZER

The elements added to forests as fertilizers occur there naturally in living and dead vegetation and in the soil. Fertilization attempts to increase plant production by providing essential elements such as nitrogen, phosphorus and potassium in forms more available to rapidly growing trees than those already present. There are many forms in which the various fertilizer components are partitioned within the forest ecosystem.

Urea is the most common nitrogen fertilizer applied in the Pacific Northwest. A coarse, granular "prill" form is used to minimize drift and maximize crown penetration during aerial application. Although urea is very soluble, it is rapidly hydrolyzed to ammonia, which is adsorbed by soil particles or taken up by living plants as ammonium ion. Urea is thus an important form of nitrogen in stream water for only a few days immediately after it has been applied. Under abnormally dry conditions Watkins and Strand (1970) reported that as much as 46% of the added urea may be lost directly to the atmosphere as ammonia, and it is therefore usually applied when temperatures are low and rain will carry it into the soil.

Under conditions of normal aeration and moisture, much of the urea is gradually converted to nitrate by soil baceteria. In this form, nitrogen is rapidly taken up by growing vegetation. Nitrate not taken up by plants remains in equilibrium with ammonium and organic nitrogen which can also be utilized by plants. Toxic amounts of nitrate in soil are unknown in forests (Miller, 1974). Any excess of nitrate not assimilated by plants may be leached into streams, however, due to its high solubility. Free-to-grow ecosystems are effective sinks for nitrate, however, and leakage is minor. Under poorly drained circumstances, especially where a high water table prevails, denitrifying bacteria may result in significant production of nitrite, which is a form much more toxic to fish.

Ammonia is added directly in some fertilizer forms or can occur as a breakdown product of urea. Ammonia is a form of nitrogen with substantial toxicity to fish under certain circumstances. Only unionized ammonia is toxic to fish and QCW describes this form as important only in warm, alkaline waters. Forest streams are usually neutral or acid (strongly acid in southeastern and north central U.S.) and are relatively cool. The range of total ammonia concentrations resulting from forest fertilization in the Pacific Northwest does not approach the critical level set by EPA.

Phosphates are rapidly adsorbed by several common soil colloids, such as clays and oxides of iron and aluminum. In this state they are only very slightly soluble, and leaching through soil becomes insignificant. Most soils have the capacity to tie up in this manner many times the quantity of phosphate which foresters are likely to apply (Powers *et al*. 1975). Growing vegetation is normally able to take up phosphate as fast as it becomes available. There are some soils of low surface adsorption capacity in which phosphate can remain in solution subsequently subject to removal by leaching. Phosphorus fertilization is now widely practiced on soils with low phosphate retention capacity and on those with very high phosphate fixing potential in the southeastern United States (Pritchett and Gooding, 1975).

PESTICIDES

Mobility of pesticides in soil depends on solubility, adsorption and persistence. Water movement is the driving force. In order for a silvicultural chemical to reach surface or ground waters by passing through soils it must be relatively soluble in water, resistant to adsorption by soil and organic matter particles, and sufficiently persistent to endure until it enters the water. Currently registered silvicultural pesticides do not combine these necessary characteristics.

A few examples may illustrate how this chain of requirements works in practice. DDT, although one of the most persistent pesticides, was detected only in the upper few inches of soil in Canadian forests which had received aerial applications for many years (Yule, 1970), because of its very low solubility. Cacodylic acid, on the other hand, is very soluble, but is adsorbed so tightly to mineral soil that it fails to penetrate more than a few inches (Norris, 1974). The organo-phosphate insecticides and carbaryl do not persist long enough to be leached through soil in significant concentrations. Phenoxy herbicides are degraded as well as adsorbed by soil particles. Picloram is the only herbicide which is both sufficiently soluble and persistent to allow measured concentrations to reach stream water in subsurface flow, and these concentrations are very low.

Routes of Chemical Movement Into Water

Aerially applied forest chemicals are initially distributed among the four sectors of the forest shown in Figure 5: air, vegetation, forest floor, and water. The proportion entering each compartment depends on the nature of the chemical and carriers applied, method of application and environmental factors during and immediately following application. Ground application is unlikely to cause direct contamination except through washing of equipment or spillage.

In order to influence water quality directly, silvicultural chemicals or their end products must reach a lake, stream or swamp (Figure 5). This occurs principally by direct application to the water while applying the chemicals, inadvertantly or otherwise. Surface movement of recently applied chemicals into the nearest stream by overland flow is another potential route for contamination. Leaching of dissolved chemicals through the soil to a water table and from there to the nearest surface water, is both a less direct and less likely path to pollution. Erosion of soil which has previously been treated with pesticide can also carry chemical constituents to water.

Distribution of pelleted fertilizer will result in maximum placement on the forest floor, with little if any in the other three sectors. Spraying of liquids from aircraft, on the other hand, may result in significant atmospheric losses, possible drifting, potential water contamination and reduced quantities reaching the forest floor. As little as 25 to 40% of a low-volatile ester herbicide has reached its immediate target area when applied from a fixed-wing aircraft. Similar percent recoveries have been reported for aerially-applied insecticides (Argauer et al., 1968; Tarrant et al., 1969). Recent developments in drift control technology have decreased losses substantially.

Pesticides reaching the litter layer, or forest floor, are subject to degradation at varying rates. Norris (1970) has investigated degradation of four herbicides widely used in the Pacific Northwest. In red alder

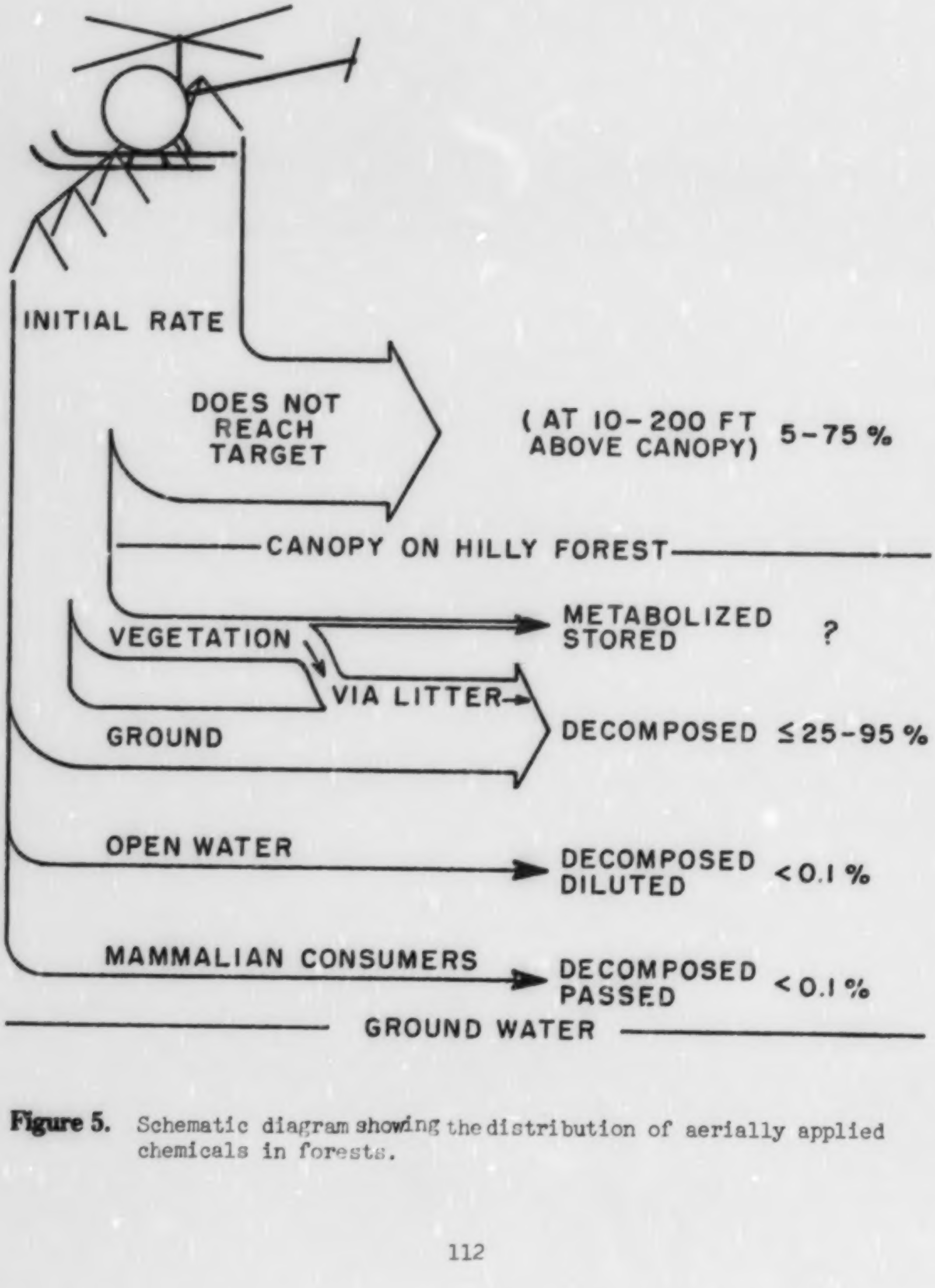
INITIAL RATE
DOES NOT REACH TARGET
(AT 10-200 FT ABOVE CANOPY) 5-75 %
CANOPY ON HILLY FOREST
VEGETATION
METABOLIZED STORED
?
VIA LITTER
GROUND
DECOMPOSED ≤25-95 %
OPEN WATER
DECOMPOSED DILUTED
<0.1 %
MAMMALIAN CONSUMERS
DECOMPOSED PASSED
<0.1 %
GROUND WATER

Figure 5. Schematic diagram showing the distribution of aerially applied chemicals in forests.

forest floor material he found that 94 percent of the 2,4-D and 80 percent of amitrole added were degraded in 35 days, while breakdown of 87 percent of the 2,4,5-T required 120 days. Only 35 percent of the picloram was degraded in 180 days.

Comparison of recovery rates among various formulations of 2,4-D in red alder forest floor material revealed that approximately 25% of the triethanolamine, isooctylester and solubilized acid forms were degraded in 15 days (Norris, 1970). More than 50% of the pure acid form was broken down in the same time.

Degradation of 2,4,5-T from red alder forest floor material was approximately 85% in 120 days, regardless of application rate. Breakdown of the phenoxy herbicides is largely due to micro-organisms. Amitrole is decomposed by chemical means as well.

That portion of an herbicide which is not decomposed on the forest floor and washes into the soil is quickly adsorbed on to soil organic material in the surface layer of forest soil. Herbicides held by organic matter in this way are very resistant to leaching and continue to be decomposed by micro-organisms.

Reported instances of significant stream contamination by forest insecticides have been the result of direct application to streams or atmospheric drift. Low rates of application and low soil mobility make it very unlikely that forest insecticides would pass through forest soils in amounts sufficient to result in damage to aquatic biota.

Much of the forest fertilizer lost in drainage water has resulted from chemicals falling directly into stream channels within the treated area. Numerous studies have shown that concentration of dissolved fertilizer (urea) in streams within the treated area is highest immediately after application (Moore, 1975). The size of this application-related peak concentration in drainage water is reduced when care is taken to avoid dropping fertilizer directly into stream channels. Due to the short-lived nature of these peak concentrations and the small percentage of any potential treatment area occupied by flowing water, a very small fraction of the total fertilizer applied is lost in this manner; nearly all is in the form of urea at the time of application.

Urea fertilizer is converted to nitrate, ammonium ion and organic nitrogen soon after application. That some nitrate constituent can eventually pass through soil into surface waters is shown by a secondary peak concentration (nitrate), which usually occurs when soils become saturated for the first time after fertilizer has been applied (Figure 6). Most of the fertilizer loss in the Pacific Northwest occurs in this form because slightly elevated nitrate levels in solutions may last for several weeks or months. Leaching and stream discharge rates associated with warm fall rains are apt to be higher than when fertilizer was applied (Moore, 1971).

Influence of Application Method on Stream Contamination

Direct contamination of streams occurs through various routes, the importance of which differs with method of application. In general, aerial application has the greatest potential for uncontrolled movement of falling droplets into open water. This is the logical result of widespread use, and a substantial component of fine droplets usually resulting from conventional nozzles operated 50 or more feet aboveground. This diffuse movement of small quantities of material usually results in some deposit in water near the

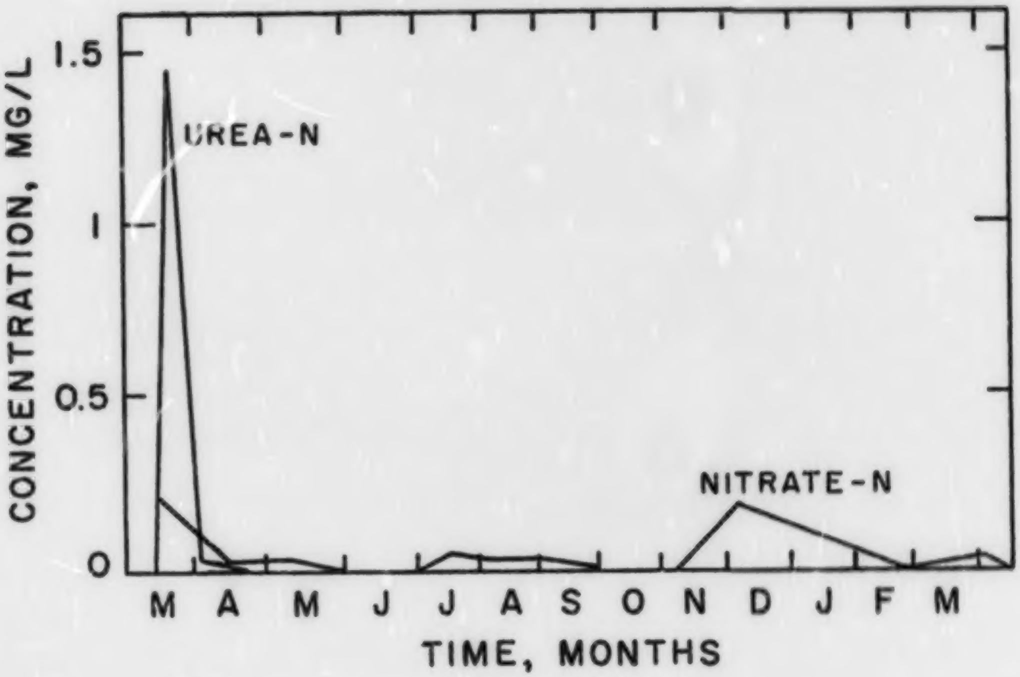

Figure 6. Graph showing representative fluctuations of urea-N and nitrate-N concentrations in streamwater following fertilization of a coniferous watershed in western Oregon with 200 lbs/acre (224 kg/ha) of urea (Fredriksen, Moore and Norris, 1975).

operation, whether detectable or not. These low-level contaminations may be decreased by modifying both equipment and the manner of use. Aerially applied pesticides are usually loaded either at an airport or on an elevated heliport some distance from open water. There is therefore little chance for contamination through mishap while loading. Leaving streamsides untreated prevents large droplets from contaminating. Contamination from this source is of low level in most instances.

Ground application embodies a different set of problems, leading presumably to different patterns of contamination. Ground applicators are more likely to be mixing and handling technical concentrates near water supplies where spillage poses a contamination hazard. Users of injectors carry the concentrate on their persons, and pose the possibility of contamination if they fall while crossing streams. In general, however, most of these postulated problems are subject to control through training and supervision.

The control of aerial contamination may entail technical modifications according to the nature of the application job. Aircraft normally apply relatively low volumes of spray per unit area. Volumes range from about one pint per acre (one liter per hectare) to 20 gallons per acre (187 l/ha). The volumes greater than 3 gallons per acre (27 l/ha) are generally used for herbicides requiring volume for penetration or suspension of wettable powders. Insecticides usually use low-volume applications.

As volume per acre decreases, the droplets must decrease in size if coverage is not to be diminished. Because small droplets remain suspended in air longer than large ones, they are more subject to drift, turbulence and movement away from the swath centerline. This is an advantage for uniform coverage between streams, but it poses a contamination problem in proximity to open water.

Fine droplets occur in almost any spray pattern. It is therefore difficult to eliminate them altogether. The proportion of volume in small droplets is substantially controlled by spray boom pressure, nozzle technology and the use of various thickeners. The degree to which fine droplets are minimized determines the proportional decrease in contamination with increasing width of streamside buffer strips. Fines are kept to a minimum by: 1) reducing boom pressure, 2) increasing orifice size, 3) orienting nozzles into the air stream, 4) using specialized boom and nozzle designs, 5) minimizing use of straight oil in spray mixtures, and 6) thickening the spray mixture by addition of various foaming agents, thickening polymers or inversion of emulsions.

The proportion of fine droplets determines certain features of the application deposit rates across a swath. In general, a given aircraft will produce a wider effective swath with fine sprays than with coarse atomization, and there will be fewer gaps or peaks within the swath. The outer edges of the swath that overlap with the next swath or extend into an untreated zone also carry a higher deposit, whereas a coarse spray has a sharper edge to its pattern. Even apart from drift, a pattern with fine droplets must therefore be farther away from open water than a coarse spray to achieve a given level of pollution control.

For a given volume of spray an increase in droplet size is accompanied by a decrease in coverage. If this is to be done without a decrease in effectiveness, higher application rates of pesticide may have to be used. There are therefore incentives for optimizing droplet size to meet control requirements and reduce risk of contamination. Ideally, this should be done by creating uniform droplets of the smallest size compatible with expected air movement and flying height.

The above brief review of application technology is not intended to be an in-depth review of the technical literature. Readers interested in technological aspects of controlling droplet distribution are referred to a substantial body of literature in the agricultural engineering area of application systems. This literature will be sampled below to review the use of application technology for determining swath widths and influence of buffer strips on deposits of chemicals.

Total swath width in an aerial spray is considerably wider than effective swath width (ESW). The latter is defined as the maximum width covered by contiguous successive swaths spaced so that deposit deficiencies are not observed at the points of overlap. Actually, most points within a sprayed area are covered by a swath of nominal application rate plus the "tails" of several other swaths (Isler and Maksymiuk, 1961; Isler and Yuill, 1963). These tails help to minimize skips in application pattern, but are the principal source of stream contamination apart from direct flyover. The amount of such deposits can be estimated from research on swath cross-sections, and from the limited field measurements of cumulative deposits.

The deposits outside the effective swath width have been described for Stearman aircraft flying 50 feet above a target (Isler and Yuill, 1963). Using an aircraft calibrated to deliver one gallon per acre in a 100-foot ESW with 150 microns mass median diameter (MMD), they observed that the deposit 100 ft. (30 M) from the centerline were one-fifth those at 50 feet (.15 M), and one tenth the nominal application rate. This relation was nearly constant for several arrangements of booms, and nozzles within booms. Recovery rate of total spray ranged from 60 to 90 percent within a double swath width. Extrapolation of these data suggests that a buffer strip of one full swath width would expose a stream to a range of deposit varying about five percent of the nominal rate. Isler and Maksymiuk (1963) observed a similar distribution of droplets of unspecified size applied at the same rate by helicopter. The fixed-wing tests were with water; helicopter tests were done with oil.

Some of the most definitive data from operational high-volume herbicide sprays (10 gallons per acre) were obtained from unpublished results in an Idaho aerial brush control operation. Mr. Paul Gravelle, Potlatch Corporation monitored herbicide deposits downwind of a helicopter application of phenoxy herbicides in a water-oil invert emulsion. His findings are shown in Figure 7, in which the deposits are plotted as the function of distance from the edge of a major (200-acre) spray project. Deposits were recorded downwind when wind velocity during the project varied between 1-10 mph. Two major patterns emerged from his data: 1) deposits were less than five percent fifty feet downwind of the project edge, and 2) virtually all deposits outside the project were of droplets $\leq$ 250 microns mass median diameter.

Gravelle's data indicate that important gains to be made from buffer strips are limited to the first 50 feet, beyond which there is a very low incidence of deposit, varying little with additional distance. The low level deposits observed could probably have been largely eliminated with a nozzle system that minimized droplets below 250 microns. In view of the dependence of insecticide applications on low volumes and small droplet sizes, the need for wider buffer strips can be visualized.

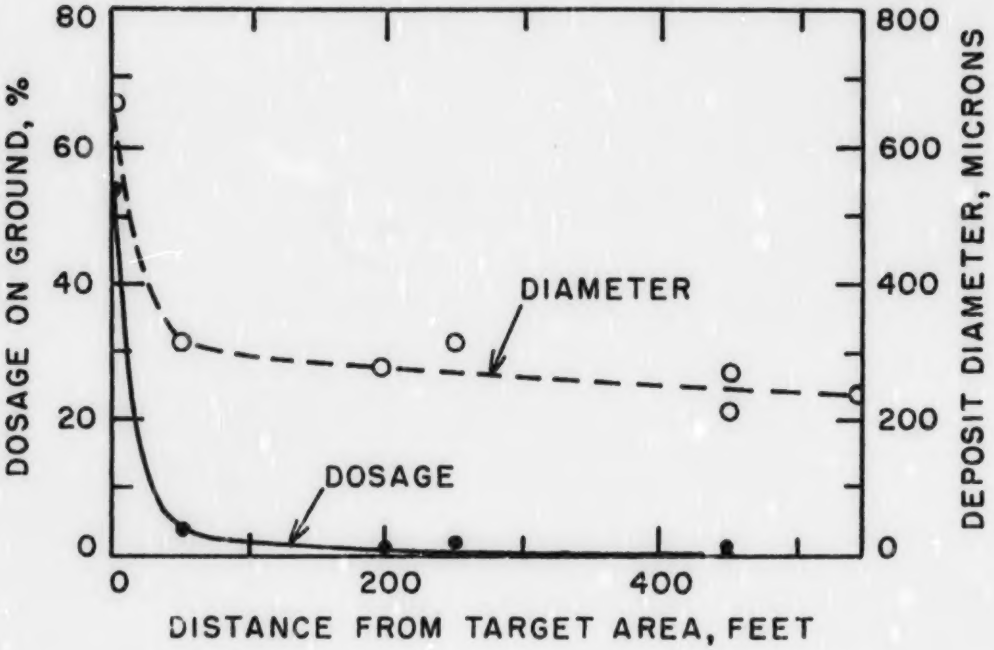

Figure 7. Percentage of nominal dosage reaching the ground, and mass median diameter of an herbicide deposit outside of and downwind from a 200 acre spray project, applied by helicopter at 10 gallons per acre water-oil emulsion in 1-10 mph crosswind. (From Gravelle, 1976, unpublished data, Potlatch Corporation, Lewiston, Idaho.)

Environmental Factors Affecting Appearance of Silvicultural Chemicals in Water

Nature and distribution of precipitation is perhaps the most widely recognized environmental influence on non-point water pollution. Stewart et al. (1975) have emphasized the primary role of rainfall in their comprehensive summary of factors affecting potential non-point water pollution from agricultural sources. Although the amount of erosion from forest land is much less than that due to agriculture, precipitation is still a dominant influence. Torrential rainfall soon after application of chemicals to forest lands is obviously more apt to result in polluted runoff than low intensity, intermittent rains which begin several weeks after a similar application.

Length and steepness of slope has less influence on pollution from forested watersheds than from agricultural lands because little or no surface runoff takes place in forests, except where soils have been severely disturbed (Dyrness, 1969).

Perhaps a more important environmental factor than slope is the nature and distribution of ground and surface water. Forest waters are most likely to show elevated levels of silvicultural chemicals in nearly level areas where a water table approaches the surface and small streams or swamps are numerous. Such contamination can occur either because of difficulty in avoiding direct application to surface water or because chemicals reach ground water as soon as they penetrate the soil.

Forest soils provide a large safety factor with respect to water pollution by silvicultural chemicals. Those pesticides which persist long enough to be washed from vegetation and through the forest floor material are quickly adsorbed onto the surface of minute soil particles, where a large variety of micro-organisms are capable of decomposing most of them. Organic debris in streams has a similar tendency to remove solutes from water.

The surface area on which chemicals may be adsorbed in soils is indeed large and occurs on both organic matter and clay particles. The organic matter content of forest soils commonly ranges between 50,000 and 500,000 pounds per acre (56,000 and 560,000 kg/ha), which provides a surface area of

4,000 to 40,000 sq. mi./acre of soil (25,000-250,000 Km^2/ha), if one assumes an average of 500 m^2 per gram of soil organic matter (Bailey and White, 1964).

The specific surface of clay is somewhat more variable, but may range higher, and the clay content of forest soils is commonly several times that of organic matter. It would not be at all unusual then, for a forest soil to contain internal surface area more than double the values cited above. In addition to surface area, clay and organic matter has a negative electric charge which enables it to attract those chemicals which are positively charged, e.g. ammonium ions and cacodylic acid. Values at the lower end of the range given above are to be expected in the sandy soils of the southeastern U.S., while those at the higher end would be most common in coastal areas of the Pacific Northwest.

Exceptions to the general effectiveness of soil adsorption for immobilizing forest chemicals are of two kinds. Very soluble materials such as nitrate can be leached from soils if there is abundant rainfall when both vegetation and soil organisms are inactive or have been disturbed. Repeated use of persistent chemicals such as DDT, will allow them to accumulate to some degree in soils where application inputs occur faster than microorganisms can destroy them. These chemicals are prevented from leaching out of soil, but are still subject to water transport by erosion, if soil is disturbed.

Patterns of Appearance of Silvicultural Chemicals in Water

Figures 6 and 8 illustrate the essential similarity among concentration curves reported for silvicultural chemicals in waters draining treated

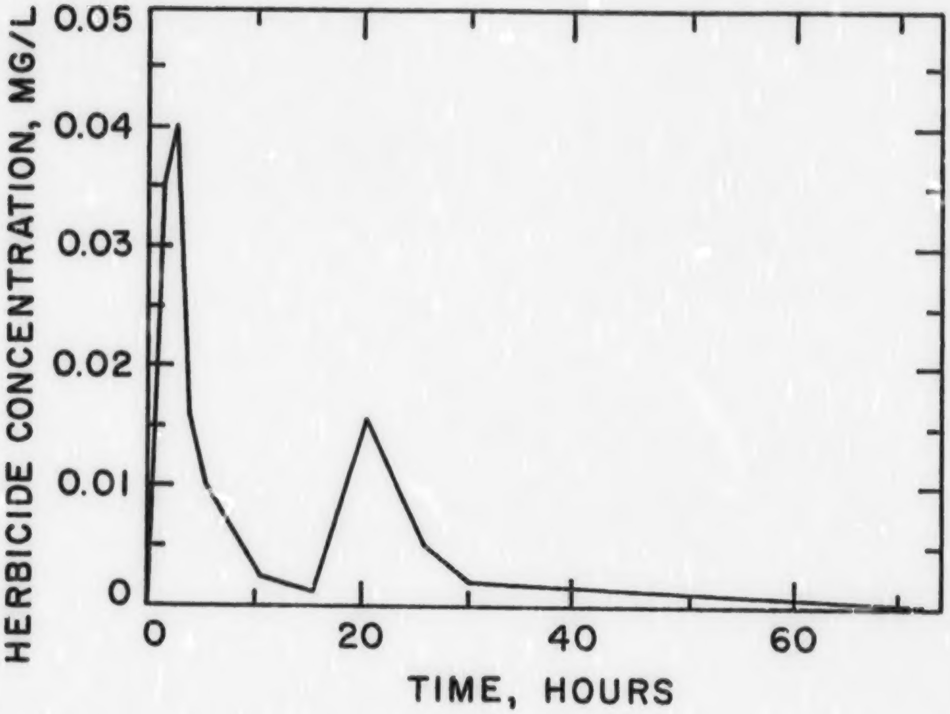

Figure 8. Graph showing a representative pattern of pesticide concentration in streamwater after spraying 10 percent of a forest watershed in 4 separate portions (Norris, 1967).

areas. Levels tend to be highest immediately following application of the chemicals (several hours to several days), then quickly drop to much lower concentrations, from which there is a gradual decline over a period of days (herbicides) or months (fertilizer). The amount of time involved is influenced by velocity of the streams concerned and regularity of the curve is influenced by precipitation. Alternation of wet and dry periods, for instance, may cause significant concentration fluctuations in chemicals of some degree of mobility.

Maximum reported concentrations following forest fertilization are 27 and 44 mg/l of nitrogen as urea (Cline, 1973; Burroughs and Froehlich, 1972). In both instances, small streams were not protected by untreated buffer strips. In the latter case, all of the watershed was treated. These values are close to theoretical expectations for a stream one foot deep receiving the rate applied (200#/acre of N) (224 kg/ha). An average peak value for watersheds receiving corresponding treatment is approximately 1/10 of those given above. Where only a fraction of a watershed was fertilized, and buffer strips were excluded, the maximum urea-N value was commonly less than 1 mg/l (Moore, 1975).

Peak ammonia-N concentrations for unbuffered streams ranged between .35 and 1.4 mg/l, while buffered and partially treated watersheds gave corresponding values between .01-.16 mg/l. The highest of these values is well below those cited as harmful to freshwater aquatic species in QCW, at expected conditions of temperature and pH. Peak values for nitrate-N in streams lacking buffer strips was 1.8-2.1 mg/l as opposed to 0.4-.17 mg/l in partially treated watersheds with buffer strips.

Nitrate concentrations vary widely in natural streamflow. Brown _et_ _al_. (1973) and Miller (1974) have examined the variation in nitrate levels in streams subjected to various perturbations in the nitrogen-rich forest soil systems of the Oregon Coast Range. Fredriksen (1970) has reported similar data from a less rich group of watersheds in the Oregon Cascades. All these reports identify considerable variability in native nitrate concentrations unrelated to application of fertilizers. With rare exceptions, native water fertility exhibits greater variability than was reported by Moore in his review of existing water impact data relating to fertilization (1975).

Data on water quality changes from forest fertilization with phosphates is limited. Those studies which have been completed indicate little or no direct effects. Sanderford (1975) reported a slight decrease in actual phosphate concentrations after fertilizing a loblolly pine plantation in the North Carolina piedmont. Compensating increases in stream flow resulted in slightly higher loss of phosphorus per unit area. Since only one-eighth of the watershed was fertilized, the increased phosphorus loss could not be positively attributed to the fertilized area.

Data from the Carolina Coastal Plain and from northern Florida indicate that phosphate levels in water draining those forested areas are very low, and are not measurably influenced by fertilization in "bedding" operations. This monitoring was done after applications of 0-49-0 fertilizer, ranging from 50 to 200 pounds per acre (56 to 224 kg/ha) applied as part of large industrial rehabilitation projects, all by ground equipment (personal communication, Mr. George Dissmeyer, U.S. Forest Service, Atlanta, Georgia).

Application of approximately 100 pounds per acre (112 kg/ha) of phosphate to coastal pine forests in western Florida resulted in small and irregular changes in phosphate concentration of local streams. Post-treatment ranges did not exceed the previously established normal range (Avers and Scott, 1976). One of the nine watersheds monitored showed a significant increase in phosphate concentration (from .03 to .06 ppm) for about one month after fertilization. The authors indicate a high probability that this increased concentration resulted from fertilizer being spread on open waters due to their intricate pattern in this drainage. Approximately one half of one percent of the applied phosphate was accounted for in stream flow.

Dr. William Pritchett, of the University of Florida, Gainsville, (personal communication, April 30, 1976) described phosphates, applied in the soluble form, as increasing the concentration of soil solution phosphorus to levels above 1 ppm for several months in flatwood soils (low in phosphorus retention capacity). However, most of these soils have spodic horizons, which are capable of fixing high levels of phosphorus, and there appears to be little danger of loss of phosphorus through the ground water. In order to minimize this possibility, less soluble forms of phosphate (ground rock phosphate) are recommended in the flatwoods. In his opinion, applied phosphate does not appear to be a significant pollutant when used in the flatwoods at ordinary rates of application.

Season of application influences the potential for toxic hazard of pesticides. Water velocity varies from season to season, and spawning of sensitive species is seasonal. The broad array of herbicides and seasons of plant sensitivity as well as low toxicity permit a broad range of low-hazard vegetation control. The limited season of target insect sensitivity, however, is a strong constraing on flexibility of insecticide use.

Relation Between Concentration of Chemical in Water and Biological Activity

The general relations between chemicals and aquatic organisms are dependent on concentration and duration of exposure. Other factors in the aquatic environment can interact to influence the exposure/response relationship. Fertilizers and pesticides react in different ways, and have different effects on ecosystem structure and function. These groups of chemicals will therefore be discussed separately in this section.

FERTILIZERS

To review the pattern of entry, aerially applied fertilizers usually appear in water as urea, nitrate or phosphate. Of the various forms of nitrogen, nitrate is the principal mobile form, and is the form most often encountered in forest watersheds other than the urea which happens to fall in open water. For practical purposes the other forms of nitrogen can be ignored.

Concentrations of nutrients in water influence stream productivity. The time interval over which concentration is expressed translates into a total quantity of nutrient passing a given point in the stream. In the absence of toxic quantities, the effect of increasing fertility over a period of time is to increase productivity of both resident and drifting plants during the period of elevated concentration. Aquatic plants are good scavengers of nutrients, and the elevated nutrient concentration cannot be expected to last as water moves downstream. The nutrients can move downstream, principally as increased biomass of algae, if not harvested by fish. In cases of long residence in impoundments or ponds, this can give rise to an algal bloom, with possible subsequent harmful effects on dissolved oxygen concentrations as plants die.

Major increases in aquatic plant biomass require sustained increases in nutrient levels. The finding that nutrient input from forest fertilization occurs as pulse contaminations (Moore, 1975) is basic to an evaluation of the likelihood of adverse effect on aquatic life. It may be postulated that a graphic representation of such chemical fertilization on biomass and productivity of aquatic systems would take the form illustrated in Figure 9.

In Figure 9, it will be observed that a brief change in productivity does not have a prolonged effect on ecosystem dynamics. A pulse of fertilizer passing a point in a stream contributes to a brief increase in productivity, which leads to the accumulation of an increment of biomass. As soon as the pulse passes, excess accumulation ceases, and the only result is a somewhat higher standing crop. Because of the general nutrient economy of free-responding ecosystems, it may be anticipated that the added increment of nutrient in the local pool will equilibrate back to its original level very slowly. The brief nature of the increased period of productivity suggests that there is not likely to be a substantial change in ecosystem structure associated with the occurrence of a pulse of fertility. In the event of a low-level, but prolonged increase, e.g. the nitrate in Figure 6, the pattern of impact on productivity would presumably follow the dashed line in Figure 9.

The effect of a pulse of fertility in a forest stream system will likely decrease with increasing distance below the fertilization project. Pulses of contamination in water change with downstream movement. Studies with various chemicals, including dyes and herbicides, suggest that downstream movement decreases concentration, and increases duration of detectable concentration (Norris, 1971; Muirhead-Thomson, 1971). This pattern presumably would occur with nutrient contamination, but the effect of contamination with nutrients would be lost even more quickly than with pesticides. The rapid loss in effect would result from scavenging of nutrients

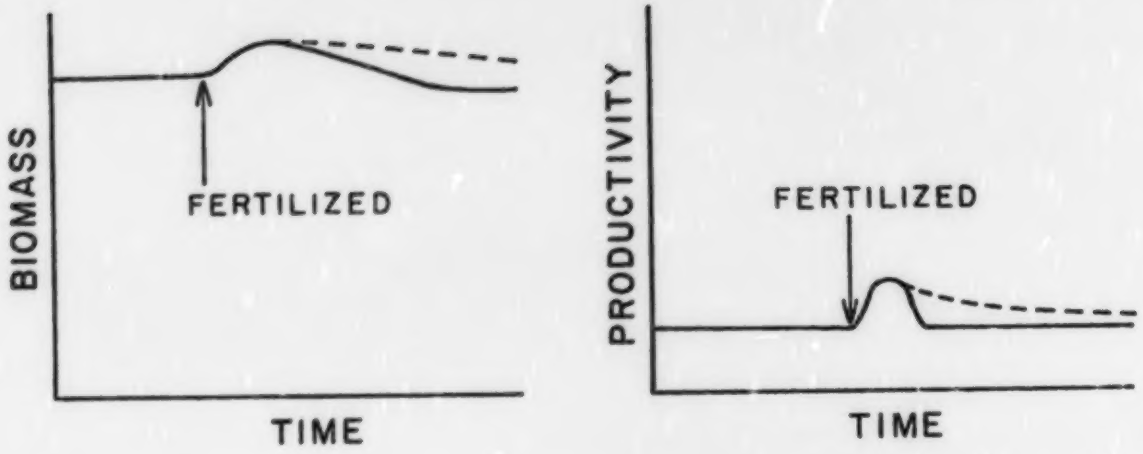

Figure 9. The postulated effect of fertilization on stream biomass and productivity.

from the water by roots and aquatic plants, by dilution and by attenuation from the native nutrient pool to which ecosystems have already adjusted themselves.

Water quality effects of forest fertilization on streams within the area treated depend largely on concentration of fertilizer components in the water. Lakes or estuaries which receive these streams, however, are capable of biologically accumulating substances in solution or suspension. Consequently, the total amount of fertilizer which is lost from a watershed over a period of time may have as much significance for eutrophication in impoundments and estuaries as the concentration in drainage water at any given time does within the forest being fertilized.

Forest ecosystems are noted for the efficiency with which they retain nutrients. Several authors (Fredriksen, 1972; Bormann et al., 1969; and

Schrieber et al., 1974) have reported that forest watersheds retain much of the nutrients added in rainfall. The net result is that levels of N and P tend to build up in forests and forest soils with the passage of time. The forms in which these elements occur in soils and vegetation are not readily soluble, and they therefore pose no direct threat of increased water pollution.

It is not surprising then, that very little of the fertilizer applied to forests is lost in drainage water. Moore (1971, 1975) reported losses ranging from .17 to .25% of that applied, during one year and seven months, respectively, for coniferous watersheds in western Oregon and Washington. Those values are equivalent to approximately .5 pounds/acre/year (.55 kg/ha/yr) and are only one quarter the increased nitrogen losses reported by Fredriksen (1970) following normal logging practices in the Northwest.

Maximum frequency of nitrogen fertilization in the Northwest is presently once in five years. In the southeastern states the interval between phosphorus fertilizations is several times longer. As currently practiced only a small fraction of a forested watershed is fertilized in a given year. Presently available evidence does not implicate forest fertilization in significant eutrophication. Data are lacking, however, to determine the long term effects of wide scale fertilizer application on stream water quality.

The area loading concept of Vollenweider (1975) implies that eutrophication potential is always relative to the hydrodynamic characteristics of the water body being studied. That is, not only concentration and total amount of nutrient added, but those factors in relation to size and flushing rate of the pond or lake being considered, determine whether detrimental accumulations of nutrients occur.

As long as even small amounts of forest fertilizers find their way to water, the possibility of causing eutrophication somewhere downstream cannot be entirely dismissed. Small lakes with large watersheds, and which receive a small annual flow of water, would be the most likely to experience eutrophication if a major part of that watershed were fertilized.

PESTICIDES

Pesticides display early patterns of concentration similar to those of fertilizers. They differ in the important respect that they do not appear in subsequent runoff, however. Pulse-like patterns are similar to those of fertilizer after aerial application, but amounts involved are very much smaller. They are not likely to be superimposed on aquatic systems in which baseline concentrations may be much higher than the amount added, and they represent a new effect on ecosystems. The potential for changing ecosystem structure therefore needs to be examined. Basic to an evaluation of potential impact is a picture of the relation between observed concentrations and estimates of "no effect" levels for each major group of species exposed.

Concepts of toxicity and toxicology are discussed in Chapter 4 of this report in the development of water quality criteria. It is germane to consider here the spectrum of toxicity, however, so that a perspective can be maintained regarding impact on a complex ecosystem when one group of species is affected more than another.

A peak of pesticide contamination entering a stream can place a selective pressure on entities within ecosystems, resulting in a change in structure. If the concentration resulting from a spray operation is far enough above the no-effect level for the most sensitive group of organisms so that it causes mortality or seriously abnormal function, the effect on the structure of an aquatic ecosystem is through temporary removal of part of a stratum. The duration of this hiatus, as well as degree, influences the likelihood of an important general ecosystem response. Insecticides tend to directly affect the animal community only, especially the insects and fish: a change in aquatic plant consumers could have an effect on the aquatic plant community. Herbicides, on the other hand, have the potential for influencing plants with no direct effect on insects or fish, but major changes in plant community structure can have severe and prolonged effects on consumers. Any prediction of effect on aquatic ecosystems must therefore consider total exposure of each major component of ecosystems subjected to pesticide. An estimation of the effects on susceptible species can then be made in relation to their influence on the rest of the ecosystem. A graphical portrayal of sensitivity of groups, and the distribution of species sensitivities within groups as in Figure 3, Chapter 4 is helpful in analyzing potential impact on community structure.

The low chronicity of pesticides applied by aircraft in silvicultural practices is significant in searching for harmful effects. Organisms of various groups can be expected to respond shortly after exposure, if symptoms are to be expressed at all. If effects are sublethal, recovery can be expected without long-term impairment of function in the way of a person recovering from toxic drugs. Even if some organisms are killed, populations of survivors can be expected to recover in accordance with the carrying capacity of their ecosystem, provided the exposure is transient. If exposure were prolonged, effects of chronic exposure would express themselves in a prolonged restructuring of the system, with dominance expressed by resistant or tolerant groups, including impacted species becoming genetically adapted to the particular adversity (Muirhead-Thomson, 1971).

The characteristic decrease in peak concentrations of pesticide pollutants with downstream movement reduces the likelihood of ecosystem impact with increasing distance below the project. The probability of direct harm is therefore greatest within the operating unit, and close to the downstream edge. This probability can be expressed in terms of the degree and duration of a concentration curve that lies above the threshold of effect (Figure 10).

Figure 10 illustrates a pattern common to all classes of chemical applied directly to open streams. The difference in effect is determined by the amplitude of the harmful effect level, and by the spectrum of species affected. Recalling the 24-hour maxima recommended for 2,4,5-T ($< .06$ mg/l), if the threshold of effect for a 24-hour exposure were one fifth the 24-hour LC_{50} of 1.4 mg/l for the most sensitive species, or 0.28 mg/l, then the herbicide would have no effect either in the area treated or downstream. If the material were an insecticide with a threshold of .01 mg/l, for aquatic insects, but 0.1 mg/l or higher for all other groups of organisms, contamination at 0.02 mg/l would produce a short term of injurious effects to insects and their predators within the treated area. These effects would largely be confined to reaches within and close to the area treated. An insecticide with a threshold of 0.0003 mg/l for rainbow trout would clearly cause serious mortality among trout and related organisms substantially below the lower monitoring point, if introduced at similar concentrations.

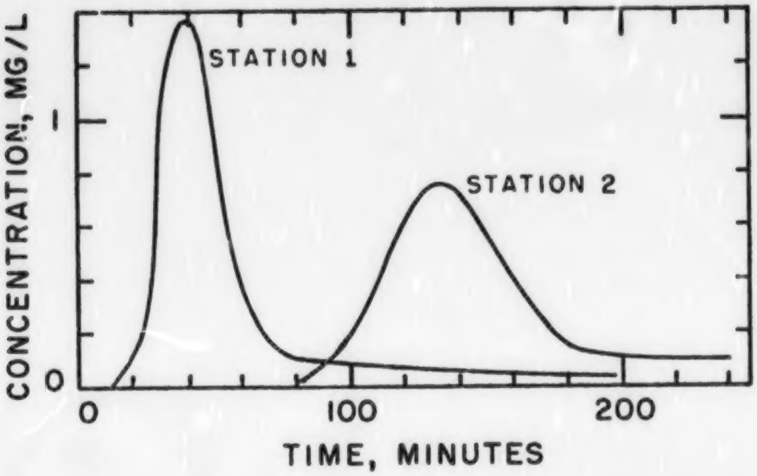

Figure 10. Comparison between chemical (dye) concentrations in stream water at the area of application (station 1) and a point downstream (station 2). Adapted from Muirhead-Thomson, 1971. If threshold of effect on the most sensitive organisms were 1 mg/l, there would be no effect discernible at Station 2.

Interaction Between Chemicals and Stream Environment

The aquatic environment exerts a substantial influence on the potential for harmful effects from chemical pollution. Among the more important factors are streamside cover, water depth at the point of contamination, stream bottom characteristics, stream velocity, degree of aeration, suspended sediment, temperature, chemical composition and acidity, dilution from seepage and tributaries and sensitivity of the aquatic community.

Stream depth within a project determines the concentration in water of any broadcast application that falls into the water directly. One pound of chemical per acre produces a maximum concentration of 0.367 mg/l in water one foot deep. The same rate of application places a maximum concentration of 0.183 mg/l in water two feet deep, but 1.468 mg/l in water three inches deep. Interception by plant cover can prevent much of this material from reaching the water surface.

Stream bottom character plays a role in two important and unrelated ways. The bottom is an important site for adsorption of chemicals; bottom configuration also affects turbulence and velocity. Stream bottoms vary considerably in roughness and surface area per unit of volume flow. If principles developed for soil are valid for stream bottoms (Bailey and White, 1964), streams with rough bottom areas have a greater potential for tying up pesticides in solution than do those with smooth bottoms. In particular, surfaces coated with organic matter have very great adsorptive capacity. Streams with very rough bottoms, high turbulence and large amounts of organic surface have the maximum capacity for reducing concentrations of chemicals in water passing through. This condition is commonplace in streams draining western and northern forests, in particular.

The process of adsorption has the important function of reducing total chemical in solution, hence peak concentrations, as a load of contaminant moves downstream. This process shortens the interval during which concentrations are likely to remain within a harmful range. The process of adsorption is reversible, however, and adsorbed materials remain in an equilibrium concentration between surface and solvent. During periods of high concentration, material is taken out of solution, but during periods of low concentration, it is slowly released. Some chemicals, such as amitrole, are complexed irreversibly, and do not reappear (Bailey and White, 1964; Norris *et al.*, 1966).

Stream velocity influences both the opportunity for adsorption of a chemical within a given reach of stream and the duration of exposure of organisms within that reach to potential harmful levels of pollution. Adsorption is a time-dependent process. Equilibration between a solution and a surface occurs over an extended period, during which a change in the concentration of chemical in water can result in a change in rate of adsorption. A decrease in solution strength to very low levels may result in desorption. The rate of desorption has not been reported as sufficient to maintain a detectable quantity of several pesticides during the period after the initial contamination peak has passed (Norris, 1968).

Flow velocity plays a major role in the rate and manner in which a pulse of contamination passes through an aquatic system. Fast-moving streams with smooth bottoms move a concentration peak downstream rapidly without rapid mixing of contaminated and uncontaminated water. Concentration peaks can remain near maximum for some distance downstream. The effect of this pattern is to sustain the levels at which harm can occur, but to confine exposures to short intervals. Fast-moving streams with substantial roughness and high ratio of volume in pools to volume in riffles, however, attenuate peaks rapidly. This type of stream prolongs the period during which chemical can be detected, but reduces the period during which harmful concentrations are present in any part of the stream system except in the zone of entry. Spreading out the concentration peak gives a maximum opportunity for degradation mechamisms to function at subthreshold contamination levels. Ponded water, however, may show elevated concentrations for extended periods even with chemicals that are normally degraded rapidly. Malathion has been found to persist at toxic levels in a pond for 14 days (Mount and Stephan, 1967), and Norris (1967) observed phenoxy herbicides to be unusually persistent in slow or stagnant water. Mount and Stephan, however, observed that the relatively more toxic butoxy ethanol ester of 2,4-D was quickly hydrolyzed to the acid under these conditions, and hazard was considerably reduced even though the acid remained detectable. It appears that toxic hazard is usually inversely affected by stream velocity, but that chemical transformation can decrease hazard even in slow water.

Biodegradation of pesticidal chemicals generally entails metabolism by many classes of organisms. Metabolism is influenced by both temperature, chemistry and aeration of water. The literature is weak regarding the principles of pesticide degradation in fresh water, and some of these effects will have to be postulated from voluminous data gathered in other systems.

Bailey and White (1964) reviewed a substantial body of soils data which indicated that adsorption is favored by low temperatures and by oil soluble formulations. These conditions also suggest low rates of metabolic activity and inaccessibility of pesticide to micro-organisms. These combinations of conditions favoring adsorption do not appear to stimulate rapid degradation, but do decrease the availability of chemical for producing harmful effects in water. Muirhead-Thomson's summary (1971) indicates that pesticides other than chlorinated hydrocarbons are least likely to produce toxic effects on fish at low temperature. Increasing temperature would tend to desorb the chemical and increase microbial activity in the presence of adequate oxygen and an energy source. Throughout the normal range of temperature, mechanisms of either physical or biological nature remove chemicals from solutions, but toxic hazard would appear to be least in cold-water streams.

The general importance of temperature in degradation of chemicals is a rate-related function. Those pesticides degraded slowly may be influenced proportionately by temperature as much as those broken down more rapidly, but the net effect in the stream system differs. Fast-degrading chemicals, such as the organophosphates, may degrade to an important degree during the short interval that an elevated concentration peak passes a given point. In slow-moving water, particularly, such degradation can make an important difference in the distance the peak moves before its amplitude is reduced below threshold levels of harmful effect; temperature effects may affect local hazard. A slowly degraded chemical, conversely, such as picloram, will not be degraded appreciably in the hours or perhaps days needed to move the chemical out of the stream system unless it is photodegraded.

Acidity has been known to affect transformation of pesticides to more toxic metabolites. For a classic example, the reader is referred to the transformation of trichlorfon to dichlorvos in alkaline (pH > 7.0) water in Chapter 4.

Availability of oxygen is generally regarded as an important prerequisite for energy-demanding microbial degradation of organic compounds. Aerobic conditions are not necessarily a requirement for breakdown, however. Blackman _et_ _al_. (1974) reported that phenoxy herbicides and picloram applied to flooded rice paddies at 26 pounds and 1 1/2 pounds per acre (29 kg and 1.7 kg/ha) respectively were sufficiently degraded in six weeks to grow an excellent crop of rice. Similar rates of degradation were observed on upland tropical soils, as measured by several crop species, and in forest soils. Apparently these pesticides can be broken down independently of level of aeration, but their report indicates general rates of breakdown under warm tropical conditions four to 12 times as rapid as reported by Norris (1970, 1971) under temperate conditions, again emphasizing the importance of temperature.

Suspended sediment provides an adsorption surface that is dispersed through the contaminated water. Although generally regarded as undesirable in over-all terms of water quality, the presence of an adsorption sink can reduce the availability of a pesticide in water or in soil. Peterson's studies with atrazine (1976) indicated that a bioassay indicator of the chemical in nutrient solution would produce a response at one-sixteenth the concentration of atrazine needed to produce the same response in the presence of soil. According to Bailey and White, (1964) the adsorptive capacities of various materials rank organic material as having very high capacity, followed in decreasing order by particles with decreasing surface charge properties, i.e., vermiculite, montromillonite, illite, chlorite, kaolinite and oxides and hydroxides.

In summary, environmental conditions within streams can have a major effect on duration and intensity of toxic hazard from a given contamination of water by a chemical. These may be summarized as in Table 4.

Effects of Operational Measures to Reduce Pollution

ROLE OF BUFFER STRIPS IN REDUCING POLLUTION

Earlier in this chapter we discussed several methods of reducing pollution during application of silvicultural chemicals. As of 1976, the only procedure required by law involves the physical isolation of treatments from open water, i.e., the use of streamside buffer strips. Forest Practices Acts in the states of Oregon, Washington and California contain rules relating to the application of chemicals by aircraft. The rules are the same for all chemicals. Specifically, the Oregon rules require that a buffer strip of one swath width be left untreated adjacent to Class I streams, i.e.,

Table 4. Stream factors affecting the duration and intensity of toxic hazard resulting from a pesticide contamination.

Factors Affecting Duration	Maximum Intensity
Ratio of pool to riffle volume	Stream velocity in the area (rapid flow decreases concentration if application period is prolonged)
Stream bottom roughness (rough decreases)	Depth of input water where applied (deep decreases)
Temperature (high decreases)	Riparian cover in project (interception decreases)
Adsorptive capacity of bottom (high exchange capacity decreases)	
Stream velocity (high decreases)	
Organic matter in stream (high decreases)	
Riparian cover below project (open, shallow streams subject to photodegradation)	

those large enough to support runs of anadromous fish or valuable for domestic use (Oregon, State of, 1972). Enforcement of this rule went into effect after considerable experience had been gained in water quality monitoring without regulation.

There are some data that bear on the effectiveness of buffer strips. Comparative research results relating to buffered versus unbuffered streamside applications are lacking, and direct comparisons have not been published elsewhere. The data base is far from precise, but there has been considerable opportunity to observe relative concentrations of herbicides in water in analogous spray operations before and after the Oregon Forest Practices Act was implemented.

Herbicide applications in western Oregon prior to 1972 resulted in highly variable concentrations in water. As a rough rule of thumb, maximum concentrations of phenoxy herbicides observed in stream water were somewhat below 0.001 mg/l per kilogram of herbicide applied per hectare for each percentage of watershed treated. Watersheds treated entirely with two pounds per acre of 2,4,5-T, (2.2 kg/ha) active ingredient basis, would be expected to show less than 0.2 mg/l 2,4,5-T in water at the point of maximum contamination by this formula. This rule of thumb may not apply directly in flat forested areas with low stream velocity. Such areas tend to have deeper streams, hence higher dilution rate. There may be a higher ratio of open water surface to land area, however, leading to greater total pesticide input. Data are not available for validation. Concentrations of phenoxys as high as this were never observed with the treatment of entire watersheds, however. The single instance of substantial underestimate by this rule of thumb was associated with a "worst case" example. Treatment of a narrow riparian salmonberry brush problem utilized amitrole on a long straight segment of stream in the Oregon Coast Range. The aircraft made repeated passes along the stream in the bottom of a deep canyon, the stream was at low flow stage in July, and the water was only a few inches deep and moving slowly. The resulting maximum concentration of amitrole after a nominal rate of application of 2 pounds active ingredient per acre was 0.8 mg/l; one mile downstream it was briefly detectable at 0.008 mg/l (Norris *et al*., 1965). Obviously, the concentration of spray in the immediate vicinity of the creek had the effect of focusing the contamination. Norris and Moore, (1970) reported concentrations of picloram and 2,4-D in a puddle under a power line. They found concentrations of 0.8 mg/l 2,4-D and over 0.1 mg/l picloram more than a month after application. The chemicals had been applied as water-soluble amine salts, six and one and one-half pounds active ingredient per acre (6.6 and 1.7 kg/ha), respectively. Obviously, a crude rule of thumb as given must be qualified by the layout of the project.

Monitoring results after applications reported since implementation of buffer strip rules have shown a reduction of herbicide peak concentrations. The degree of reduction from the one-swath rule is difficult to ascertain because of variation in rules among land managing agencies. The Siuslaw National Forest, U.S. Forest Service, observes a swath width of 100 feet along each side of Class I streams to comply with the Oregon Forest Practices Act. Using data from Gravelle (Figure 6) a deposit of less than 2 percent of that of a direct overflight would be expected with a high volume helicopter application of herbicide. Data from stream monitoring prior to initiation of this procedure can be compared to those gathered later in a more or less direct comparison. In general, the range of concentrations since implementation of buffer strips has been reduced to peak levels between one-third and one-half the former concentrations. Feeder stream concentrations pre-1972 seldom exceeded 0.05 mg/l; post-1972 concentrations have seldom exceeded .02 mg/l (L. A. Norris, 1976, personal communication). In view of Gravelle's findings, it is likely that the lower-than-expected control of pollution is attributable to the difficulty of maintaining an even 100 foot strip along an irregular creek.

EFFECT OF APPLICATION METHOD

The effectiveness of buffer strips in minimizing pollution is dependent on the droplet sizes being applied adjacent to the strip, basic swath width and degree of precision in control of the aircraft. As discussed earlier, movement of the spray across the buffer strip is inversely related to droplet size. Measures to eliminate fine droplets increase the effectiveness of a given isolation. Several of these are capable of minimizing droplets less than the 250 microns identified by Gravelle's data as the maximum size capable of major movement in a 1-10 mph wind. This approach is probably restricted to herbicides because of the low feasibility of reducing coverage of insecticides.

Aircraft guidance along irregular waterways renders difficult the observance of precise buffers. Fixed-wing aircraft are less able to maneuver than helicopters and larger aircraft are less maneuverable than small craft. Larger aircraft also apply wider swaths, presumably with proportionally wide deposits of droplets beyond the effective swath margins. There is therefore incentive for avoiding large fixed-wing aircraft applications near waterways, and for using helicopters where possible adjacent to any buffer zone. Even with helicopters, maneuvering along a crooked watercourse can increase contamination. Turns away from a creek cause helicopter rotor wash to direct an air blast containing spray toward the creek. Buffer strips along a meandering stream thus need to be wide enough to preserve the nominal protection distance with a minimum of sharp turning.

Guidance of the aircraft in relation to buffer strips has traditionally been up to the pilot. It is not practically possible to mark edges of buffer zones on the ground continuously except in certain circumstances. The one instance in which this is feasible is that of foliage herbicide application to a hardwood forest or brushfield. The boundary can be marked by injecting trees along the edge of the buffer zone with herbicide so as to create a line of defoliating trees visible from the air. Various devices have been described for intermittent marking of spray unit boundaries under more general circumstances (Maksymiuk, 1975). At least one electronics manufacturer (Motorola) has designed a precision guidance system for aircraft that permits adherance to a ground-controlled vector within one meter. Although this is interesting for maintaining accurate swath placement in the general application, it is probably not adapted to irregular edges.

SOURCES OF TROUBLE

Examination of the "worst cases" relating to the above amitrole incident, and application to slow-moving or still water with no cover, permits some tentative conclusions about procedures associated with potentially serious contamination. Every instance of concentration that exceeded the general rule of thumb was observed where shallow, slow-moving water was treated directly, and perhaps repeatedly, within an extended stretch of stream or pond. Even these instances have produced contamination levels below concentrations shown to have caused mortality among the most sensitive aquatic fauna (see Appendix, Table 1). Under unusual circumstances, Picloram concentrations in irrigation water could be harmful to certain crops regardless of a one-swath buffer, depending on crop and distance downstream to irrigation intake. Comparable contamination by numerous of the insecticides would cause severe damage (see Appendix).

Insecticides have not been studied in relation to specific effects of buffer strips. Kerswill's report of New Brunswick operations (1967) indicated that the size of aircraft used for application influenced the degree to which open water could be avoided. He reported substantial fish kills in the Miramichi River when open water was included in DDT spray patterns. Mortality of fish began when large aircraft replaced smaller Stearman craft that had been spraying smaller blocks laid out in irregular patterns to conform with landform. This report did not specify the width of buffer that was used with the Stearman aircraft, but did substantiate the importance of not applying DDT to open water. He also reported that damage to the fishery ceased altogether when phosphamidon was substituted for DDT, again without specifying the pattern in which the phosphamidon was applied. Reported levels of 0.1 mg/l phosphamidon in water suggests that no buffer strips were used, and indeed that water could not have been excluded from direct application.

The general substitution of selective, non-persistent and non-accumulating insecticides for chlorinated hydrocarbons has reduced fish mortality. Muirhead-Thomson (1971) indicated that mexacarbate, fenitrothion, malathion and phosphamidon all were much less toxic to fish than DDT, and were short-lived. Kerswill (1966, as reviewed by Muirhead-Thomson, 1971) observed that Atlantic salmon demonstrated a one-hour lethal threshold concentration of 220 mg/l, phosphamidon, more than 2,000 times as high as the 0.1 mg/l normally observed as maximum in aerial spray operations, with complete recovery. Kerswill (1967) also reported that the application of two half-rate treatments of DDT produced the same degree of injury to fish as a single full dosage. Elson and Kerswill, (1966) reported little effect on fish when a combination of DDT and phosphamidon was used, with phosphamidon being used at one-half pound per acre within buffer strips along streams. Thus, there is evidence that buffer strips can be treated without injury to sensitive fisheries, but they should not be treated with chlorinated hydrocarbons. The role of biological insect control agents for this purpose has not been elucidated with respect to water quality, but should be safe in the conventional sense.

RESOURCE COSTS IN POLLUTION CONTROL

Even if using buffer strips illustrates a clear and reasonable pattern in modifying concentrations of toxic substances in streamflow, buffer strips also may have an effect, sometimes adverse, on the non-aquatic resources. The acreage included in buffer zones tends to be lower slopes, with deep soils having favorable water regimes. Buffer zones 100 feet wide require 25 acres per mile of stream. The typically favorable site quality of these acres for timber suggests that timber resource productivity within these zones is proportionately higher than the relative area of land surface included in the buffer strips.

Productivity within buffer strips may or may not be compromised by lack of treatment, depending on the nature of the problem and proposed treatment. Buffer strips left in areas treated for site preparation with herbicides remain largely out of timber production if they are in a non-productive state initially. They remain productive as big game habitat, possibly at a level inferior to the areas sprayed with phenoxy herbicides, if cover is above browse height. In a recent report, Newton (1976) has described research with the shade tolerant western hemlock indicating that successful reforestation with such species is possible with minimum site preparation. This report does conclude, however, that heavy overhead competition limits success, and this method is principally a means for reducing intensity of herbicide use, rather than eliminating all use. Areas excluded from release herbicide sprays are likely to sustain heavy losses of growth from rapidly encroaching alder and salmonberry in coastal areas of the Pacific Northwest. Losses can be measured in terms of lost investment in plantations or in unrealized stumpage revenue in the future. The extent of losses has not been calculated precisely. Release sprays are not normally scheduled unless suppression is imminent or already present. On such areas, it is estimated that loss can amount to 25-100 percent of a plantation with 1976 potential value of $200-1,000 per acre. The total value and proportion of value lost are generally lower elsewhere in the United States. Crude estimates can be made on the basis of potential productivity. Where there is clear evidence of adverse effects of buffer strips and no observable gains in condition of aquatic systems resulting from their use, the merits of buffers are open to question.

Effects of buffer strips on insect populations remain conjectural. Observations made during the 1974 outbreak of Douglas-fir tussock moth indicated that populations were commonly decreasing at the time DDT was applied, and the 200-foot-wide buffer strips did not carry high surviving populations. Moreover, streamside habitats do not normally carry the heaviest populations. An insect species with less dramatic population fluctuations and with females which travel considerable distances, e.g. the eastern spruce budworm, could reinfest sprayed areas from buffer strips. There are thus apparently areas where buffers are useful and consistent with insect management regimes, but there may be exceptions. In view of the potential risk of injury to aquatic systems from insecticides, however, any decision to abandon buffers carries considerable risk of contamination by these materials.

Non-herbicidal programs in vegetation management are in widespread use. They are sometimes recommended as means of reducing environmental impact.

Hand felling of brush and hardwoods, and mechanical scarification are the most common. Of these, hand felling presumably has no direct effect on water quality, but scarification can have serious repercussions. Froehlich (1974) observed that there are only limited periods when machine use does not result in serious compaction and loss of infiltration capacity on some forest soils. Steep topography aggravates the runoff problem, and many agencies do not scarify on slopes greater than 30 percent. Resulting runoff after devegetation causes soil losses and siltation problems even on gentle topography. Reports of soil loss and siltation are legion in relation to agricultural operations. Dissmeyer (1973) reported that soil losses in the southeastern forest region were severe following site preparation methods that expose and disturb soil in areas of high-intensity precipitation. Methods that do not disturb the soil apparently are associated with water quality similar to that from undisturbed forests. Thus, it is clear that the substitution of mechanical methods for herbicides reduces the concentration of herbicide in water, but tends to increase the silt load and turbidity substantially. Successful hand-cutting methods prevent both siltation and herbicide contamination. Subsequent herbicide treatment is often required to achieve adequate control of sprouting species, however.

Aerial application of endrin-treated conifer seed invariably deposits some endrin in creeks. Amounts are extremely small, and endrin is not known to have had measurable impact on stream biota, despite the occasional occurrence of detectable quantities in water (Moore _et al_., 1974). The effect of endrin in forest streams has not been investigated in detail, however, and the assumption of negligible effect is an extrapolation of laboratory data. Substitute practices include the choices of applying untreated seed or hand planting. Both choices eliminate the addition of endrin to water. The use of untreated seed generally results in seeding failure (Bever, 1953), whereas hand planting increases success in reforestation considerably. The latter pattern is so well established that hand planting has supplanted aerial seeding for most reforestation efforts, despite its higher cost.

Indirect methods of wildlife damage control include the use of herbicides in habitat management. Borrecco (1973) reported successful management of small mammal populations on an experimental basis by controlling certain species of plants with herbicides. Vegetation control had the simultaneous effects of reducing animal predation on seedlings and reducing vegetative competition to seedlings. Cost of such use of herb control is greater than for endrin treatment, and the practice is normally used in conjunction with planting rather than seeding. The benefits of such weed control in plantation establishment are substantial (Newton, 1970). The herbicides used to this end are used at application rates roughly 500 times as great as those of endrin, but with less toxic hazard.

Proposals have been made for the development of management systems involving generally low impact, but preserving the advantages of chemicals. Newton has proposed (1975 letter to Oregon State Forestry Department) that Oregon Forest Practice rules permit the application of herbicides within buffer zones when forestry-registered materials are available that control streamside species, and are also registered for stream and ditchbank control. The herbicide ammonium ethyl carbamoyl phosphonate has such registration status, and is particularly effective on salmonberry and vine maple in the Pacific Northwest. This practice could likely have an impact on streamside vegetation and its ability to shade the stream channel. The application of this approach can properly be decided on the basis of debris and temperature criteria in the absence of toxic hazard. Ammonium ethyl carbamoyl phosphonate is not applied in oil.

Siltation resulting from chronic suppression of vegetation in Christmas tree plantations can be slowed by vegetation management programs involving herbicides. Newton proposed to the Northwest Christmas Tree Growers Association (1974 letter) that a program be developed in which short residue herbicides be used in conjunction with the use of grass or nitrogen-fixing legume cover crops. There is ample research and operational data suggesting that such practices are highly effective, but the appropriate herbicides have not been registered for this purpose.

Soil fertilization practices, except for those involving Christmas tree production, have traditionally relied on aerial application. Urea nitrogen fertilizers are applied as prilled products that distribute uniformly. Buffer zones along creeks can be narrow without substantial application to open water. Phosphate applications have utilized a considerable amount of rock phosphate, applied as a dust. This material can move in atmospheric drift in patterns less controllable than those obtained with prills. Dr. Pritchett, of the University of Florida at Gainsville (personal communication, 1976) has indicated that rock phosphate dust undoubtedly can settle in open water during aerial fertilization operations. He has observed, however, that phosphate concentrations are characteristically low in the southern Coastal Plain and Flatwoods area. It is likely that application with ground equipment at the time of planting would reduce losses to even lower levels, while enhancing availability to planted seedlings. Partly acidulated and prilled ground rock phosphate has been developed by TVA. When available, its use will reduce drift during application.

A relatively recent development in forest fertilization experimentation has been spraying liquid urea-ammonium nitrate solution directly on foliage. Since this approach does not attempt to add chemicals to forest soils, the probability of loss to streams should be minimal. Preliminary results indicate that stream contamination due to foliar fertilization is less than from conventional urea application, as pellets. Since foliar rates of application have been somewhat lighter (150#/acre of N), smaller fertilizer losses may be attributable to that factor (unpublished BLM report, Roseburg, OR, 1975, R. Miller and S. Wert). Increased wood production per unit of nitrogen added presently favors the foliar procedure. If fertilizer costs continue to rise, this method may be more commonly used in the future, especially when means are developed to avoid the foliar damage which can occur at present rates of application.

The general impacts of the silvicultural chemical practices as currently applied without buffer zones are summarized in Table 5. Several non-chemical practices are listed for purposes of comparison.

Table 5. Summary of effects of aerial application of chemicals and alternative silvicultural practices on water quality.

Practice	Chemical Used	Pollutant Pattern	Duration of Measurable Pollution	Group Most Likely to be Affected by Pollution if Any
Fertilization	Urea	Brief elevation of urea and low-level ammonium concentrations. Slight later elevation of nitrate.	Urea limited to immediately after application. Some elevation of nitrate from first fall rains after dry summer.	None known. Low possibility of injury to fish from NH_3 in warm water of high pH.
	Phosphorus	Brief elevation of phosphate concentration.	Limited to flooding runoff period.	None known.
Forest Site Preparation				
Chemical	Amitrole	Spike concentration*.	1-7 days.	None known.
	Ammonium ethyl carbamoyl phosphonate	"	"	"
	Atrazine/Simazine	"	"	"
	Dalapon	"	"	"
	Picloram	Spike concentration*, followed by very slight contamination in runoff or seepage.	Spike is brief. Seepage may continue several months in range ≤ 5 ppb.	Irrigation water users (potatoes, tobacco, legumes).

Table 5. (continued)

Practice	Chemical Used	Pollutant Pattern	Duration of Measurable Pollution	Group Most Likely to be Affected by Pollution if Any
	Phenoxys	Spike concentration*.	1-7 days	None known.
Non-Chemical				
Scarification:				
Steep ≥ 30% slope	None	Turbidity with storms; mass soil failures.	During storm flow, as long as soil is devegetated.	All, some severely.
Gentle 5-30% slope	None	Turbidity with storms. Most serious in areas with wet summers.	"	Aquatic systems, especially spawning fish, potable users.
Flat ≤ 5% slope	None	Turbidity with storms in wet summer areas only.	"	Potable users.
Scarification & Chemical	2,4,5-T	Turbidity plus chemical associated with silt.	"	Aquatic systems, potable users.
	Atrazine	"	Storm flow only, as long as soil is devegetated. Atrazine prolongs period of exposed soil.	"
Fire	None	Turbidity with storms, mild to severe, depending on steepness, intensity of fire and rains.	Storm flow, only while devegetated.	None to severe on fish spawning beds.

Table 5. (continued)

Practice	Chemical Used	Pollutant Pattern	Duration of Measurable Pollution	Group Most Likely to be Affected by Pollution if Any
Fire Plus Chemical	2,4,5-T and/or Dinoseb	Mild turbidity. Spike* of herbicide.	Storm flow only until site revegetates.	None to severe on fish spawning beds.
Insect Control				
Biological	None	Not known.	Not known.	None observed.
Chemical	Carbaryl	Spikes* in many streams coalesce into prolonged river pattern of low concentration.	1-7 days in feeder streams; 1-3 days longer in river.	Aquatic insects; effects no more than 2 weeks. Non-cumulative.
	Diazinon	"	"	None known.
	Disulfoton	"	"	Aquatic insects; effects no more than 2 weeks, non-cumulative.
	Endosulfan	Spikes* in many streams coalesce into prolonged pattern of low concentration in rivers; persistent chlorinated hydrocarbon may appear in very low concentrations later.	Persistent. May be problem with food chain species for extended period.	Fish extremely sensitive. Also aquatic insects. Impact likely even with buffer strips.
	Fenitrothion	Spike* in many streams coalesce into prolonged river pattern of low concentration.	1-7 days in feeder streams; 1-3 days longer in river.	Possibly aquatic insects, low impact unless applied directly to water.

Table 5. (continued)

Practice	Chemical Used	Pollutant Pattern	Duration of Measurable Pollution	Group Most Likely to be Affected by Pollution if Any
Insect Control (cont.)	Guthion	Spike* in many streams coalesce into prolonged river pattern of low concentration.	1-7 days in feeder streams; 1-3 days longer in river.	Aquatic insects very sensitive - impact likely, but brief even with narrow buffer strips.
	Lindane	Not used aerially; not found in water.	None	None
	Malathion	Spikes* in many streams coalesce into prolonged pattern of low concentration in rivers.	1-7 days in feeder streams; 1-3 days longer in rivers.	Aquatic insects very sensitive, some impact likely, but brief, without buffer strips.
	Phosphamidon	Spikes* of low concentration in many streams coalesce into prolonged pattern of low concentration in rivers.	"	Aquatic insects if water sprayed directly.
	Trichlorfon	"	"	"
Rodent Control				
Seed Coat	Endrin	Brief spike*, very low maximum.	Persistent, but levels too low for toxic hazard.	None known.

Table 5. (continued)

Practice	Chemical Used	Pollutant Pattern	Duration of Measurable Pollution	Group Most Likely to be Affected by Pollution if Any
Herbicide	Triazines, 2,4-D Dalapon	Brief spike*.	1-7 days.	None known.

* "Spike" concentrations are defined as those which diminish to less than ten percent of maximum within 48 hours.

POLLUTION CONTROL GUIDELINES

The pollution control strategy outlined in this document is directed primarily toward maintenance of forest water quality. A secondary goal is preservation of flexibility in achieving silvicultural objectives within those goals. Development of pollution control strategy has thus far emphasized determination of practices having the most potential for harming water quality and review of procedures minimizing adverse effects. This analysis has included a broad array of silvicultural practices, many of which are applied throughout the United States.

Silvicultural practices have been analyzed for their impacts on water quality from the standpoints of chemical entry into aquatic ecosystems or water used for drinking or irrigation. Chemical concentrations in water have been proposed at which significant harm is not likely to befall either aquatic ecosystem structure or productivity. Especially stringent criteria have been developed where unusually sensitive aquatic species or user groups have been identified. Larger margins of safety have been developed for major river systems than for smaller streams. Alternative practices have been evaluated both from the standpoint of water quality and effectiveness in reaching silvicultural objectives. Non-chemical water quality impacts have been included in a comparison of alternatives.

Water pollution from use of chemicals in forestry is not a widespread problem. Levels of pollution that cause measurable harm to water users or aquatic ecosystems occur infrequently. Problems that do exist are restricted to the persistent herbicide, picloram, and to the use of insecticides on large areas. At present, virtually all aerially applied herbicides and fertilizers are used under the supervision of professional foresters and technical specialists who are aware of the chief potential hazards of misuse. Quality control is generally good in field operations, and water monitoring has reflected a lack of accidents involving spills in watercourses. The potentially more hazardous insecticide applications are usually coordinated public-private operations with considerable prior organization and review. These use patterns are apparently consistent with maintenance of forest watershed quality goals. The exceptions are the principal targets for additional controls.

Priorities for Pollution Control

The goal of eliminating serious water quality impacts from the use of forest chemicals has largely been achieved by restricting the general use of organochlorine insecticides, and by avoiding direct application of insecticides to open water. There is some remaining hazard to aquatic insects and other food species from the more toxic of the organophosphate insecticides, and from the remaining occasional use of endosulfan. Even though their use has not had major obvious effects on fish, the scale of operations suggests that reduction of their potential for injuring aquatic systems with insecticides should receive the highest priority. Principal needs are adequate buffer strips and special application techniques near the water.

Second priority is given to the maintenance of productivity in streamside buffer strips, and in many other areas, by means other than mechanical vegetation control. Herbicides are available that are effective and safe in these areas. Their use can maintain productivity without causing siltation or other problems relating to soil damage. Presently registered materials can be used effectively to the water's edge, and others in developmental stages can be used with negligible impact on water quality with comparable or greater effectiveness in competition control. Procedures for establishing stands of conifers in residual brush cover need to be developed so that less intensive vegetation control is needed. Part of this task is the development of improved application precision. The removal of buffer strips from the productive forest land base is a land use question that can be considered independently of pollution only in the presence of non-polluting technology.

Third priority is given to the training and licensing of applicators so as to minimize direct contamination from ground equipment and aerial applicator ground crews. Particular attention is needed for avoidance of spills, dumping of wash water and back siphoning from spray tanks.

Fourth priority is given to the problem of picloram and dicamba residues in water used for irrigation. This is principally a physical placement problem in relation to both proximity to water, and stream distance between herbicide projects and irrigation users with sensitive crops. This priority is given in anticipation of future problems.

As a general silvicultural goal, there needs to be development of management systems that prevent the occurrence of the problems on which pesticidal chemicals are normally used. Progress toward that end could reduce future need to rely on chemical expedients. Epidemic levels of pests, both animal and plant, are often outgrowths of management that failed to include adequate consideration for pest prevention. Little can be done about past errors, but future silvicultural operations can prevent development of many problems on the scale at which they now occur. This goal is inextricably intertwined with the direct pollution control priorities.

Rules for Application of Chemicals in Silviculture

Table 6 provides a list of forest management practices involving the application of chemicals, and outlines the rules for buffer strip treatment and monitoring so as to meet the goals of this program. Methods used to reduce impact of chemicals (Priority I) include designation of buffer zones of widths in accordance with the potential hazard posed by the chemical. The rationale behind recommendations for buffer strip widths is based on the earlier described 20-fold decrease of contamination with each herbicide swath width away from the stream and a five-fold decrease per swath of low-volume insecticides with winds less than 5 mph. Based on experience with various pesticides, the proposed criteria for water concentrations (Chapter 4, Table 3) will be met with a margin of safety when registered rates of application are applied as recommended. In those exceptions where buffer strips are defined in terms of absolute width, the problem is the physical movement overland or through the soil in subsurface flow, a group of processes not affected by application technology.

To achieve the second priority, meeting forest production goals without compromising water quality, emphasis is given to the identification of practices that have adverse impacts near water, and substituting less harmful practices. Exceptional conditions under which untreated buffer zones are recommended are identified so that unnecessary loss of productivity can be avoided.

Monitoring

Monitoring will be needed to insure that the recommendations are a) being observed, and b) effective in maintaining water quality. Monitoring responsibility for validation of practices will be the responsibility of state and federal water resources agencies, and operational quality control will be the responsibility of the operator. Monitoring by users will be necessary on a limited scale to provide a record of the consequence of the chemical activity at the point of maximum potential trouble. The intensity of monitoring is specified in Table 6. Specific treatment of samples is outlined in procedures compiled by Freed _et al_. (1971), of which a portion germane to water is reproduced below:

> Sampling Points
>
> Selection of an appropriate sampling station is important since the value of the information obtained is only as good as the sample collected. The sample should be representative of the volume of water passing the sampling point, and it should be possible to collect a sample without stirring up bottom sediments or kicking surface debris into the stream. When the treatment unit lies adjacent to the stream to be sampled, the sampling point must be downstream of all small side channels flowing from the treated area. At the same time, however, we want to sample the stream as close to the lower boundary as possible so that the samples will represent the maximum concentration of chemicals to which aquatic organisms may have been exposed.
>
> Control samples are normally collected at the same sampling station prior to spraying, but in some situations it will be possible to obtain these samples upstream from the spray unit. In either case, the sampling point should not be subject to contamination by aerial drift during the sampling period. In critical situations such as spraying brush above a water intake for a fish hatchery or water supply, sampling stations should be established near the intake.
>
> Collection of Samples
>
> Before the project is begun, appropriate sample containers must be obtained. The type and size of sample, the container

and conditions for storage and transport should be verified with the analytical laboratory. If herbicides like 2,4-D, 2,4,5-T or picloram, are used, samples should be taken in glass containers and treated with sodium hydroxide (NaOH) to prevent loss of any herbicide residue that may be present. Amitrole is highly soluble in water and stable so treatment with sodium hydroxide is not needed. It should be collected in glass containers. For fertilizer, more intensive background sampling is required and the sample containers and mode of preservation differ. It is essential to check with an analytical laboratory or the chemist who will do the analysis to determine container and preservation requirements. Samples should be representative of the total volume of water flowing past the collection point. For small streams, collect a grab sample at the lower end of a straight, narrow length of channel carrying a steady flow of water. On larger streams, the samples should be taken near the center of the channel at a depth of 2-4 inches. The individual collecting the samples must not have any herbicides or other contaminants on his hands or clothing and the sample containers must also be free of contamination. These precautions are extremely critical because of the sensitivity of analytical methods. As a general precaution the person collecting the samples should not have any other contact with an active spray project.

Each sample must be clearly identified and all pertinent information correctly and completely recorded on a tag or label securely attached to the container. In addition to assigning an identifying number, the attached tag or label should show the date and time collected, location, weather conditions since time of application, and name of collector. Other information that may be recorded is the rate of application, chemical formulation used and size of area treated.

Timing of Collection

The number of samples collected and the timing, or sampling interval, will depend in part on the particular project being monitored. Following is an example of the sampling sequence that might be used for a chemical brush control project which is not near a highly sensitive area.

Hours after spraying is begun:

1. Control (prespray)
2. 15 minutes to 1 hour
3. 3 hours
4. 24 hours
5. 48 hours
6. 72 hours

Timing of the collection of sample number 2 depends on the distance between the lower unit boundary and the sampling point. If this point is immediately below the unit, sample 2 should be taken within 15 minutes after spraying is started. If the sampling point is downstream some distance below the unit, collection of sample 2 can be delayed. This timing also applies when the unit is sprayed in late evening to take advantage of good weather conditions. Sample 2 should be taken at the time interval indicated and sample 3 can be delayed until the next morning. In this case, samples 4, 5, and 6 would still be taken at 24-hour intervals from the time spraying started. The guiding principle is to collect samples when you expect residues to pass the sampling point.

Due to poor weather, equipment failure, or the size of the area, it is often necessary to spray a unit over a period of several days. Should this occur during a monitoring program, samples 2 and 3 should be taken each day that spray is applied. Samples 4, 5, and 6 then would be taken at 24-hour intervals after the last application on the unit. When the treatment unit lies within a municipal watershed or in a watershed that supplies a fish hatchery, additional samples should be taken 5 to 10 days after application. Following rain and wind, any time within the first 3 to 4 weeks after application may require the collection of additional water samples. Many chemicals tend to be tightly held in the forest floor and soil, but it is possible that residues along the stream may move into the stream channel by subsurface or overland flow. At least one sample should be taken during and after the period of peak flow.

Transport and Storage

Sample containers, whether empty or full, should not be transported or stored with chemicals. As soon as sampling has been completed, the accumulated samples should be shipped to the laboratory for chemical analysis. If for some reason the samples are not analyzed immediately, storage conditions should be verified with the analytical laboratory.

Chemical analysis for pesticide residues is an expensive proposition because of the time and equipment required. It may be desirable to reduce the cost of monitoring on some projects by compositing some of the samples and thereby reducing the number of analyses required. This can be done by combining equal parts of each of several samples taken at a monitoring point, excluding the control sample. No more than 4 or 5 samples should be included in a composite and the remainder of each individual sample should be saved in case the analytical results on the composite show that more detailed information is needed. The composite sample must be so marked and a complete identification included with it when submitted for analysis.

> Safety
>
> All pesticides and samples that may contain their residues, are potentially poisonous and should be handled accordingly. They are selected for their toxic properties toward a specific target and every effort must be made to insure that nontarget components of the environment are not adversely affected as the result of misuse or carlessness. In the case of an accidental spill or drop of herbicide into open streams, lakes, or other bodies of water, all interested persons should be notified and monitoring procedures should be started immediately.

Monitoring is an inexact procedure, and some weaknesses in present procedure need to be resolved. In particular, methods need to be developed that distinguish between dissolved and adsorbed chemicals. Monitoring recommendations include filtration for some of the more strongly adsorbed chemicals so as to determine solution levels. There is no feasible method at present for monitoring concentrations in small aquatic organisms. Monitoring procedural amendments to the above general guide are given in Table 6. Noteworthy among the deviations from Freed _et_ _al_. (1971) are the frequency of samples. Frequent samples are necessary for research toward determination of patterns of fluctuation of chemical concentration. Operational samples are feasible at lower frequency because they may be interpreted in terms of existing research data on concentration fluctuation.

Pesticide analyses are exacting and costly. It may not be necessary to have samples analyzed if there is no evidence of contamination. The samples should be taken and stored under refrigeration for at least 3 months, however. If a problem arises, samples should be analyzed by competent commercial or government laboratories equipped for the chemicals and precision needed.

Table 6. Guidelines for Applying chemicals by aircraft, and water monitoring in silvicultural practices.

Practice	Chemical Used	Minimum Distance Between Nearest Water and Center Line of Nearest Swath	Treatment of Buffer	Suggested Location and Frequency of Water Sampling
Fertilization	Urea	3/4 of an effective swath width (ESW).	Apply by ground rig.	Composite, Day 1, at potable user site, if within 1 mile downstream from project.
	Phosphorus	3/4 ESW* Exceptions: upstream from lake or impoundment.	Apply by ground rig.	None
Forest Site Preparation	Amitrole	1/2 ESW* Exceptions: within a mile of potable users, 50-foot buffer.	a) Apply by ground rig. b) Apply substitute chemical. c) Plant buffer zone with tolerant tree species.	Composite, Days 1 & 2, at potable user site if within 1 mile downstream.
	Ammonium ethyl Carbomyl phosphonate	1/2 ESW*	Can be treated.	Composite, Day 1 at potable user site if within 1 mile of project downstream.
	Atrazine	1/2 ESW* Exceptions: scarified areas, 50 feet.	Do not disturb soil within buffer zone.	None
	Dalapon	1/2 ESW	Do not disturb soil within 50 feet of creek.	None
	Phenoxys	1/2 ESW	Can be treated.	Composite, Day 1, at intake if potable user within 1 mile of project downstream.

Table 6. (continued)

Practice	Chemical Used	Minimum Distance Between Nearest Water and Center Line of Nearest Swath	Treatment of Buffer	Suggested Location and Frequency of Water Sampling
	Picloram	100 feet (200 feet when applied during period of rainfall surplus).	Can be treated with substitute chemical within prescribed limits.	Composite, weekly at irrigation user if within 5 miles of project, and crops include potatoes, tobacco or legumes. Sample after spraying, again in sequence after effective rainfall.
Forest Insect Control				
Biological	Bacillus thuringiensis	None*	Can be treated.	None
	Nuclear polyhedrosis	None*	Can be treated.	None
Chemical	Carbaryl	1 ESW* or 100 feet, whichever is greater.	May treat with biological agent.	Composite each day of spraying immediately downstream from project and above potable user, and 2 days after. Sample at water intake, if within 2 miles of project. Filter samples.
	Diazinon	1 ESW* or 100 feet, whichever is greater.	"	"
	Disulfoton	1 ESW* or 100 feet, whichever is greater.	"	"

Table 6. (continued)

Practice	Chemical Used	Minimum Distance Between Nearest Water and Center Line of Nearest Swath	Treatment of Buffer	Suggested Location and Frequency of Water Sampling
Forest Insect Control (cont.)	Endosulfan	4 ESW* or 300 feet, whichever is greater.	May treat with carbaryl diazinon, fenitrothion or phosphamidon to 1 ESW from water.	Sample as with organophosphorus insecticides, but sample also after each heavy rain for next month.
	Fenitrothion	1 ESW*	May use biological agent.	Same as carbaryl.
	Guthion	3 ESW* or 200 feet, whichever is greater.	"	"
	Malathion	"	"	"
	Phosphamidon	1 ESW* or 100 feet, whichever is greater.	"	"
	Trichlorfon	"	"	"
Rodent Control (Seeding)				
Chemical	Endrin	3/4 ESW	Can be treated by hand.	None

*For definition and discussion of ESW see pages 118 and 119.
Designation of "None" or 1/2 ESW under Buffer Strip Width implies only that buffer strip width is at the discretion of the operator, and that direct impact on water quality is not at issue. Even without a buffer strip, the aircraft should never be operated within a half-ESW of streams that are likely to have fish in them at time of chemical application. For those insecticides requiring one or more effective swath widths, the proposed buffers are for helicopters with droplet size of 200 u MMD. If droplets are smaller or large fixed-wing aircraft are used, buffers should be 200 feet plus the given swath numbers. Helicopters may be used in conjunction with large aircraft.

REFERENCES

Anonymous. 1975. Guidelines for areawide waste treatment management planning. U.S. Environmental Protection Agency, Washington, D.C.

Argauer, R. J., H. C. Mason, C. Corley, A. H. Higgins, J. N. Sauls, and L. A. Liljedahl. 1968. Drift of water-diluted and undiluted formulations of malathion and azinphosmethyl applied by airplane. Jour. of Economic Entomology 61:1015-1020.

Argyle, R. L., G. C. Williams and H. K. Dupree. 1973. Endrin uptake and release by fingerling channel catfish (*Ictalurus panctatus*). Jour. Fish. Res. Bd. Can. 30:1743-1744.

Audus, L. J. 1970. The action of herbicides on the microflora of the soil. Proc. 10th Br. Weed Cont. Conf. 1036-1051.

Avers, P. E. and Bobby Scott. 1976. Effects of forest fertilization on water quality in the Apalachicola National Forest and management implications. USDA Nat. Forest Service, In house Report. 10 p. 34 graphs.

Bailey, G. W. and J. L. White. 1964. Review of adsorption and desorption of organic pesticides by soil colloids with implications concerning pesticide bioactivity. J. Agr. Food Chem. 12:324-332.

Beavers, J. T. 1957. Removing cull hardwoods. Farm and Home Science Magazine (Tenn. Ag. Exp. Sta.) 12:6-7.

Bever, Dale N. 1953. 1948-1949 aerial seeding experiments. Oregon State Board of Forestry Bulletin 9. 22 p.

Black, H. C. and B. T. Vladimiroff. 1963. Effects of grazing on regeneration of Douglas-fir in southwest Oregon. Soc. Amer. Foresters Proc. 69-76. Boston.

Blackman, G. E., J. D. Fryer, A. Lang, and M. Newton. 1974. Persistence and disappearance of herbicides in tropical soils. Technical Supplement, Effects of Herbicides in South Vietnam. National Academy of Sciences. 59 p.

Bollen, W. B. 1961. Interactions between pesticides and soil microorganisms. Ann. Rev. Microbiol. 15:69-92.

Bollen, W. B. 1962. Herbicides and soil microorganisms. Proc. Western Weed Control Conf. Las Vegas. p. 48-50.

Borman, F. H., G. E. Likens, and J. S. Eaton. 1969. Biotic regulation of particulate and solution losses from a forest ecosystem. Bioscience 19:600-610.

Borrecco, J. E. 1973. The response of animals to herbicide-induced habitat changes. Oregon State Univ. Masters thesis. 92 numb. leaves.

Borrecco, J. E., H. C. Black, and E. F. Hooven. 1972. Response of black-tailed deer to herbicide-induced habitat changes. In: Proc. West. Assoc. of State Game and Fish Commissioners. Portland, Oregon. p. 437-451.

Brown, G. W., A. R. Gahler, and R. B. Marston. 1973. Nutrient losses after clear-cut logging and slash burning in the Oregon Coast Range. Water Resources Research 9:1450-1453.

Brown, George W. and J. T. Krygier. 1970. Effects of clear-cutting on stream temperature. Water Resources Research 6(4):1133-1139.

Bruns, V. F., J. M. Hodgson, and H. F. Arle. 1972. Response of several crops to soil herbicides in irrigation water. U.S.D.A. Tech. Bull. 1461.

Bruns, V. F., B. L. Carlile, and A. D. Kelley. 1973. Responses and residues in sugarbeets, soybeans and corn irrigated with 2,4-D or silvex-treated water. U.S.D.A. Tech. Bull. 1476.

Bruns, V. F., R. J. Dermint, P. A. Frank, A. D. Kelley, and J. C. Pringle Jr. 1974. Responses and residues in six crops irrigated with water containing 2,4-D. College of Agric. Res. Ctr. Washington State Univ. Bull. 798.

Buckman, H. O. and N. C. Brady. 1960. The nature and properties of soils. Macmillan, New York, N.Y. 567 p.

Burroughs, E. R. Jr. and H. A. Froehlich. 1972. Effects of forest fertilization on water quality in two small Oregon watersheds. U.S.D.I. Bureau of Land Management Technical Note.

Cline, Charles. 1973. The effects of forest fertilization on the Tahuya River, Kitsap Peninsula, Wash. Washington State Dept. of Ecology 74-2. 54 p.

Council for Agricultural Science and Technology. 1974. Zero concepts in air, water and food quality legislation. Report No. 31.

Courtney, K. D., D. W. Gaylor, M. D. Hogan, H. L. Falk, R. R. Bates, and I. Mitchell. 1969. Teratogenic evaluation of 2,4,5-T. Science 168: 864-866.

Courtney, K. D., J. A. Moore, D. W. Gaylor, M. D. Hogen, and H. L. Falk. 1970. Summary teratogen study NIEHS. Hearings of the Hart Subcommittee on 2,4,5-T. Washington, D.C.

Crosby, D. G. and A. S. Wong. 1976. Environmental degradation of 2,3,7,8 tetrachlorodibenzo-p-dioxin (TCDD). Submitted to Science for publication. Department of Environmental Tocicology. Univ. Calif., Davis.

Day, Boysie et al. 1975. The phenoxy herbicides. Council for Agricultural Science and Technology. Report #39. Iowa State Univ., Ames. 21 p.

Dissmeyer, George E. 1973. Evaluating the impact of individual forest management on suspended sediment. Soil Conservation Soc. of America Proceedings, Plants, Animals and Man. p. 258-264.

Dissmeyer, George E. 1974. Managing pine forests to improve water quality. In: Proceedings of Symposium on Management of Young Pines. USDA, Forest Service Southeast Forest Experiment Station, Asheville, North Carolina.

Dyrness, C. T. 1969. Hydrologic properties of soils on three small watersheds in the western Cascades of Oregon. USDA, For. Service Research Note PNW-111.

Elson, P. F. and C. J. Kerswill. 1966. Impact on salmon of spraying insecticide over forests. Adv. Water. Poll. Res. 1:55-74.

Fitzgerald, C. H., F. A. Peevy Jr., and D. E. Fender. 1973. The southern region. Jour. Forestry 71(3):148-153.

Fredriksen, R. L. 1970. Comparative chemical water quality - natural and disturbed streams following logging and slash burning. p. 125-137. In: Forest Land Uses and Stream Environment. J. T. Krygier and J. D. Hall eds. OSU Press. Oregon State University, Corvallis.

Fredriksen, R. L. 1972. Nutrient budget of a Douglas-fir forest on an experimental watershed in western Oregon. In: Research on Coniferous Forest Ecosystems. J. F. Franklin, L. J. Dempster and R. H. Waring eds. Pacific Northwest Forest and Range Experiment Station.

Fredriksen, R. L., D. G. Moore, and L. A. Norris. 1973. The impact of timber harvest, fertilization, and herbicide treatment on stream water quality. Forest Soils and Forest Land Management. Proceedings of the Fourth North Amer. Forest Soils Conf., Laval Univ., Quebec, Canada. B. Bernier and C. H. Winget eds.

Froehlich, H. A. 1974. Soil compaction: implications for young-growth management. In: Managing Young Forests in the Douglas-fir Region. A. B. Berg ed. School of Forestry, Oregon State Univ., Corvallis.

Goldberg, L. 1970. Letter to W. B. Ennis, U.S. Department of Agriculture. As quoted in anonymous U.S.D.A. report on status of knowledge regarding 2,4,5-T, 1971.

Gooley, Walter R. Jr. 1973. The ubiquitos budworm. American Forests 79 (5):46-53.

Gratkowski, H. 1967. Ecological considerations in brush control. In: Herbicides and Vegetation Management. Symposium Proceedings. M. Newton ed. Oregon State Univ., School of Forestry, Corvallis.

Gratkowski, H. J., D. Hopkins, and P. G. Lauterbach. 1973. The Pacific Coast and northern Rocky Mountain region. Jour. Forestry 71(3):138-143.

Greaves, Robert D. 1974. Some factors affecting the distribution of organic arsenical silvicides in the forest ecosystem. M.S. thesis, Oregon State Univ., School of Forestry, Corvallis. 94 p.

Hatfield, C. T. 1969. Effects of aerial spraying with Sumithion and Phosphamidon on salmon brooks in western and central Newfoundland. Can. Dept. of Fish. & Forests. Progress Rept. 33 p. + tables.

Hayes, W. J. Jr. 1967. The 90-day LD_{50} and a chronicity factor as measures of toxicity. Tox. and Appl. Pharmacol. 11:327-335.

Isensee, A. R. and G. E. Jones. 1975. Distribution of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) in aquatic model ecosystems. Environ. Sci. Tech. 9:668-672.

Isler, D. A. and Bohdan Maksymiuk. 1961. Some spray distribution and atomization tests with a helicopter. U.S. Dept. Agri. ARS Rept. 42-54.

Isler, D. A. and J. S. Yuill. 1963. Spray distribution patterns from a Stearman airplane flying at 50 feet. U.S. Dept. Agr. ARS Rept. 42-82.

Jukes, T. H. and C. B. Shaffer. 1960. Antithyroid effect of aminotriazole. Science 132(3422):296-297.

Kerswill, C. J. 1967. Studies on effects of forest spraying with insecticides, 1952-1963, on fish and aquatic invertebrates in New Brunswick streams. Jour. Fisheries Res. Bd. Canada 24(4):701-708.

Kerswill, C. J. and H. E. Edwards. 1967. Fish losses after forest spraying with insecticides in New Brunswick, 1952-1963, as shown by caged specimens and other observations. Fish Res. Bd. Canada 24(4):709-729.

Klock, G. O. 1971. Streamflow nitrogen loss following forest erosion control fertilization. Pacific Northwest Forest and Range Experiment Station USFS Res. Note PNW-169.

Koerber, T. W. 1976. Lindane in forestry, a continuing controversy. Pacific Southwest Forest and Range Experiment Station. General Technical Report PSW-14.

Leak, W. B. and C. W. Martin. 1975. Relationship of stand age to streamwater nitrate in New Hampshire. U.S.D.A. Forest Service. Northeast Forest and Range Experiment Station Research Note NE-211.

Leng, M. L. 1974. Final report. Workshop B. Environmental Impact. USDA/Dow 2,4,5-T Scientific Conference. Washington, D.C.

Likens, G. E., F. H. Bormann, N. M. Johnson, D. W. Fisher, and R. S. Pierce. 1970. Effects of forest cutting and herbicide treatment on nutrient budgets in the Hubbard Brook watershed-ecosystem. Ecol. Mono. 40:23-47.

Loomis, T. A. 1974. Essentials of toxicology. 2nd edition. Lea and Febiger. Philadelphia.

Lueschow, L. A. 1964. The effects of arsenic trioxide used in aquatic weed control operations on selected aspects of the bio-environment. M.S. thesis. University of Wisc., Madison.

Maksymiuk, Bohdan. 1975. Marking methods for improving aerial application of forest pesticides. U.S. Dept. Agr. Forest Service Res. Note DNW-262.

Maleug, K. W., G. F. Powers, and D. F. Krawczyk. 1972. Effects of aerial forest fertilization with urea pellets on nitrogen levels in a mountain stream. NW Science 45:52-58.

Mantel, N., N. R. Bohidar, C. C. Brown, J. L. Ciminera, and T. W. Tukey. 1975. An improved Mantel-Bryan procedure for "safety" testing of carcinogens. Cancer Research 35:865-872.

Marks, G. C. and T. T. Kozlowski eds. 1973. Ectomycorrhizae: Their Ecology & Physiology. New York, Academic Press. p. 1-205.

Marston, R. B., D. W. Schults, Tamotsu Shiroyama, and L. V. Snyder. 1968. Amitrole concentrations in creek waters downstream from an aerially sprayed watershed sub-basin. Pesticides Monitoring Journal 2(3):123-128.

Matsumura, Fumio. 1975. Toxicology of Insecticides. Plenum. New York.

Meehan, W. R., L. A. Norris and H. S. Sears. 1974. Toxicity of various formulations of 2,4-D to salmonids in southeast Alaska. J. Fish. Res. Bd. Canada 31:480-485.

Miller, James H. 1974. Nutrient losses and nitrogen mineralization on forested watersheds in Oregon's Coast Range. Oregon State Univ. Ph.D. thesis. 83 numb. leaves.

Miller, R. A. 1974. The chronic toxicity of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) to five selected aquatic organisms. M.S. thesis. Oregon State Univ. 62 p.

Miller, R. A., L. A. Norris, and C. L. Hawkes. 1973. Toxicity of 2,3,7,8 tetrachlorodibenzo-p-dioxin (TCDD) in aquatic organisms. Env. Health Perspectives 1973:177-186.

Moore, Duane G. 1971. Fertilization and water quality. In: Western Reforestation, Proceedings An. Meet. West. Ref. Coord. Com. West. For. Conserv. Assoc. p. 28-31.

Moore, Duane G. 1975a. Effects of forest fertilization with urea on stream water quality - Quilcene Ranger District, Wash. USDA Forest Service Research Note PNW-241.

Moore, Duane G. 1975b. Impact of forest fertilization on water quality in the Douglas-fir region - a summary of monitoring studies. Soc. of Amer. Foresters Proceeding New York, Sept. 22-26, 1974.

Moore, Duane G., J. D. Hall, and W. L. Hug. 1974. Endrin in forest streams after aerial seeding with endrin-coated Douglas-fir seed. USDA Forest Service Research Note PNW-219.

Moore, D. G. and L. A. Norris. 1974. Soil processes and introduced chemicals. In: Environmental Effects of Forest Residues Management in the Pacific Northwest. U.S.D.A. Forest Service Gen. Tech. Rept. PNW-24.

Mount, D. I. and C. E. Stephan. 1967. A method for establishing acceptable toxicant limits for fish - malathion and the butoxyethanol ester of 2,4-D. Trans. Am. Fish. Soc. 96:185-193.

Mount, D. I. and G. J. Putnicki. 1966. Summary report of the 1963 Mississippi fish kill. Trans. of N. Amer. Wildl. and Nat. Res. Conf. p. 177.

Muirhead-Thomson, R. C. 1971. Pesticides and freshwater fauna. Academic Press. London. 248 p.

Newton, M. 1964. Seedling survival and vegetative competition. In: Western Reforestation. Western Forestry and Conservation Ass'n. Portland. p. 39-42.

Newton, M. 1970. Planting trees in the Pacific Northwest. Pacific Northwest Coop. Ext. Service Tech. Bull. PNW-120.

Newton, M. 1973. Forest rehabilitation in North America: some simplifications. Journal of Forestry 71(3):159-162.

Newton, M. 1977. A test of western hemlock wildlings in brushfield reclamation. Oregon State Univ., Forest Research Laboratory Research Paper. (In press).

Newton, M., B. A. El Hassan, and J. Zavitkovski. 1968. Role of red alder in western Oregon forest succession. In: Biology of Alder (J. M. Trappe et al. eds.). U.S.D.A. Forest Service Pacific Northwest Forest and Range Experiment Station. Portland. p. 73-84.

Newton, M. and L. A. Norris. 1976 Evaluating short- and long-term effects of herbicides on non-target forest and range biota. Symposium on Biological Evaluation of Environmental Impact, Council for Environmental Quality. Ann. Mtg. Ecol. Soc. Amer. and AIBS. New Orleans. (Published also in Down to Earth 32(3):18-26.)

Newton, M. and R. W. Smith. 1976. Injecting and aerial spraying herbicides in Vermont. Manuscript submitted to Northern Logger.

Norris, L. A. 1967. Chemical brush control and herbicide residues in the forest environment. In: Herbicides and Vegetation Management. p. 103-123. M. Newton ed. Oregon State University, Corvallis.

Norris, Logan A. 1970. Degradation of herbicides in the forest floor. p. 397-411. In: Tree Growth and Forest Soils (C. T. Youngberg and C. B. Davey, eds.). Oregon State Univ. Press, Corvallis.

Norris, L. A. 1971. The behavior of chemicals in the forest. In: Pesticides, Pest Control and Safety on Forest and Range Lands. p. 90-106. Oregon State University Coop. Ext. Serv. Corvallis. J. M. Witt ed.

Norris, L. A. 1974. The behavior and impact of organic arsenical herbicides in the forest: final report on cooperative studies. U.S.D.A. Forest Service Pacific Northwest Forest and Range Experiment Station. Corvallis, Oregon.

Norris, L. A. and R. A. Miller. 1974. The toxicity of 2,3,7,8 tetrachlorodibenzo-p-dioxin (TCDD) in guppies (*Poecilia reticulatus* Peters.). Bull. Env. Contam. and Toxicol. 12(1):76-80.

Norris, Logan A. and Marvin L. Montgomery. 1975. Dicamba residues in streams after forest spraying. Bull. Environ. Contam. Toxicol. 13(1). 8 p.

Norris, Logan A. and Duane G. Moore. 1970. The entry and fate of forest chemicals in streams. In: Forest Land Uses and Stream Env. p. 138-158. J. T. Krygier and J. D. Hall eds. Oregon State Univ. 252 p.

Norris, L. A., M. Newton, and J. Zavitkovski. 1965. Streamwater contamination by herbicides resulting from brush-control operations on forest lands. Western Weed Control Conference Res. Prog. Rep. p. 35-37.

Norris, L. A., M. Newton and J. Zavitkovski. 1966. Stream contamination with amitrole following brush control operations with amitrole-T. Western Weed Control Conference, Research Progress Reports. p. 20-22.

Norris, L. A., M. Newton, and J. Zavitkovski. 1966. Chemical brush control and contamination in forest watersheds. Western Weed Control Conference Abstracts. Reno. p. 147-148.

Ogg, A. G. Jr. and L. A. Wapensky. 1969. Measurement of picloram persistance in two western U.S. soils. Abstract. Weed Science Society of America Annual Meeting. Las Vegas.

Peterson, C. A. 1976. The metabolic pathways and physiologic effects of atrazine in Douglas-fir seedlings. Ph.D. thesis. Oregon State University, Corvallis.

Pierovich, John M., E. H. Clarke, S. G. Pickford, and F. R. Ward. 1975. Forest residues management guidelines for the Pacific Northwest. USDA Forest Service Gen. Tech. Rpt. PNW-33. 272 p.

Powers, R. F., K. Isik, and P. J. Zinke. 1975. Adding phosphorus to forest soils: storage capacity and possible risks. Bull. Env. Cont. and Toxicology 14:257-264.

Pritchett, W. L. and J. W. Gooding. 1975. Fertilizer recommendations for pines in the U.S. Southeastern coastal plain. Florida Agr. Experiment Station Bull. 774. 21 p.

Ries, S. K., H. Chmiel, D. R. Dilley, and P. Filner. 1967. The increase in nitrate reductase activity and protein content of plants treated with simazine. Proceedings of the Nat. Acad. of Science 58:526-532.

Roberts, C. A. 1975. Initial plant succession after brown and burn site preparation on an alder-dominated brushfield in the Oregon Coast Range. M.S. thesis. Oregon State Univ., School of Forestry. 90 p.

Rohlich, G. A. (ed.). 1972. Water quality criteria. National Academy of Sciences Environmental Studies Board, Washington, D.C. 594 p.

Rose, J. Q., J. C. Ramsey, T. H. Weutzler, R. A. Hummell, and P. J. Gehring. 1976. The fate of 2,3,7,8 tetrachlorodibenzo-p-dioxin following single and repeated oral doses to the rat. Toxcol. Appl Pharmacol. 35:209-226.

Ross, Ralph. 1976. Unpublished presentation at the Annual Meeting Western Society of Weed Science. Portland. (Dr. Ross is Project Director, Dioxin Implementation Study, Environmental Protection Agency, Washington, D.C.)

Rudinsky, J. A. 1963. Response of Dendroctonus pseudotsugae Hopkins to volatile attractants. In: Contributions, Boyce Thompson Institute 22: 23-28.

Rudinsky, J. A. 1970. Sequence of Douglas-fir beetle attraction and its ecological significance. In: Contributions, Boyce Thompson Institute 24:311-314.

Sanderford, Sharon G. 1975. Forest fertilization and water quality in the North Carolina Piedmont. Tech. Report #53. N.C. State Forest Fertilization Coop. 42 p.

Schreiber, J. D., P. D. Duffy, and D. C. McClurkin. 1974. Sediment and nutrient losses from five pine watersheds in northern Mississippi. Agron. Abstr. 38.

Sprague, J. B., P. F. Elson, and J. R. Duffy. 1971. Decrease in DDT residue in young salmon after forest spraying in New Brunswick. Env. Pollution 1:191-203.

Stewart, B. A. et al. 1975. Control of water pollution from cropland. Volume I, A manual for guideline development. 111 p. U.S. Environmental Protection Agency, Agriculture and Non-Point Source Management Division. Report No. EPA-600/2-75-026a. 111 p.

Symons, P. E. K. 1973. Behavior of young Atlantic salmon (Salmo salar) exposed to or force fed fenitrothion, an organophosphate insecticide.

Symons, P. E. K. and G. D. Harding. 1974. Biomass changes of stream fishes after forest spraying with the insecticide Fenitrothion. Fisheries Research Board of Canada, Tech. Rept. #432.

Tarrant, R. F., D. G. Moore, W. B. Bollen, and B. R. Loper. 1972. DDT residues in forest floor and soil after aerial spraying, Oregon--1965-1968. Pestic. Monit. J. 6(1):65-72.

Tracy, H. B. and D. M. McGaughy. 1975. Aquatic monitoring of the 1974 tussock moth DDT aerial spray project in Washington State. State of Washington Department of Ecology Report DOE 75-17.

Tracy, H. B. and Bill Purvis. 1976. Aquatic monitoring of the 1975 spruce budworm fenitrothion aerial spray project in Washington State. State of Washington Department of Ecology Report DOE 76-2. 34 p.

U. S. Dept. of Interior. 1968. Water Quality Criteria. National Technical Advisory Committee, Fed. Water Pollution Control Admin.

U.S. Environmental Protection Agency. 1975. Quality criteria for water. Draft document. 501 p.

U.S.D.A. 1975. Vegetation management with herbicides. Final Environmental Statement. Siskiyou, Siuslaw and Umpqua National Forests. U.S. Forest Service, Portland.

U.S.D.A. 1975. Vegetation management with herbicides. Final Environmental Statement. Deschutes, Fremont, Ochoco and Winema National Forests. U.S. Forest Service, Portland.

U.S.D.A. 1975. Vegetation management with herbicides. Final Environmental Statement. Ozark-St. Francis National Forest. U.S. Forest Service, Atlanta. 244 p.

Vollenweider, Richard A. 1975. Input-output models with special reference to the phosphorus loading concept in limnology. Swiss Journal of Hydrology 37:53-84.

Von Rümker, Rose Marie. 1974. Production, distribution, use and environmenatl impact potential of selected pesticides. Office of Pesticide Programs, Env. Protection Agency, USDI.

Walker, C. M. 1973. Rehabilitation of forest land. J. For. 71(3):136-137.

Watkins, S. H. and R. F. Strand. 1970. Volatilization loss of ammonia nitrogen from urea applied to forest soils and forest floor materials. Agronomy Abstracts. p. 164.

Wildish, D. J. and N. A. Lister. 1973. Biological effects of fenitrothion in the diet of brook trout. Bulletin of Environmental Contamination and Toxicology 10:333-339.

Yule, W. N. and A. D. Tomlin. 1970. DDT in forest streams. Bull. Environ. Contam. and Toxicol. 5(6):479-488.

Appendix Table I. Toxicity Data For Silvicultural Chemicals

Chemical	Organism	Mammals and Birds Acute Oral LD_{50} mg/kg of Body Weight Aquatic Organisms Acute LC_{50} Mg/1	Chronic Effects	No Effect Level	Reference
Fertilizer					
Nitrate	Fish				
	Bluegill	$NaNO_3$ 200 96 hr			Trama 1954
	Bluegill	KNO_3 420 96 hr			Trama 1954
	Chinook salmon	1360 96 hr			Westin 1974
	Rainbow trout	1310 96 hr			Westin 1974
	Largemouth bass			400	Knepp & Arkin 1973
	Channel catfish			400	Knepp & Arkin 1973
Nitrite	Fish				
	Chinook salmon	.900 mg/1 96 hr			Westin 1974
	Rainbow trout	.550 mg/1 24 hr		.150 mg/1 48 hrs	Smith & Williams 1974
	Rainbow trout, flow thru.	.190 mg/1 96 hr		60 mg/1 10 days	Russo et al. 1974
		.390 mg/1 96 hr			

Appendix Table I. (continued)

Chemical	Organism	Mammals and Birds Acute Oral LD_{50} mg/kg of Body Weight Aquatic Organisms Acute LC_{50} Mg/l	Chronic Effects	No Effect Level	Reference
Herbicides					
Amitrole aminotriazole	Mammal				
	Rat	1100.			Bailey & White 1965
	Fish				
	Largemouth bass		62.5 mg/l 14 day		Bond 1960
	Bluegill	100 mg/l 48 hr	Cont. flow, all survived		Sanders 1970
	Coho salmon	32.5 mg/l 24 hr			Bond *et al.* 1960
	Crustacean				
	Crayfish			100. mg/l 48 hr	Sanders 1970
	Daphnia magna	.003 mg/l 48 hr			Sanders 1970
Ammonium ethyl carbamoyl phosphonate (K[illegible]te)	Mammal				
	Rat	24,000 mg/kg		2,200 mg/kg/day	Dupont Corp.
	Guinea pig	7,380 mg/kg		2 weeks	Tech. Inf.
	Birds				
	Mallards	> 10,000 mg/kg			
	Bobwhite quail	> 10,000 mg/kg			

Appendix Table I. (continued)

Chemical	Organism	Mammals and Birds Acute Oral LD_{50} mg/kg of Body Weight Aquatic Organisms Acute LC_{50} Mg/1	Chronic Effects	No Effect Level	Reference
Herbicides (cont.)					
Ammonium ethyl carbamoyl phosphonate (cont.)	Fish				
	Fathead minnow	> 1,000 mg/1			
	Bluegill	670 mg/1			
	Rainbow trout	> 1,000 mg/1			
Arsenicals	Mammals				
Cacodylic acid (dimethylarsinic acid)	Rat (male)	1,400		100 mg/kg, feed/d, 90d	US EPA 540-1-75-021
	Birds				
	Chicken			100 mg/kg, feed/d, 10d	
	Mallard duck	>2,000			
	Chukar	>2,000			
	Fish				
	Bluegill	16. mg/1 96 hr			

Appendix Table I. (continued)

Chemical	Organism	Mammals and Birds Acute Oral LD$_{50}$ mg/kg of Body Weight Aquatic Organisms Acute LC$_{50}$ Mg/l	Chronic Effects	No Effect Level	Reference
Herbicides (cont.)					
Arsenical	**Marine & Estuarine**				
Cacodylic acid (dimethylarsinic acid)	Pink shrimp			40. mg/l 48 hr	
	Eastern oyster			1. mg/l 96 hr	
	Daphnia magna			.001 mg/l 32 d	
	Mammals				
Monosodium Methane Arsonate (MSMA)	Rat	1800.		100 mg/kg of feed, 90 d.	US EPA-540-1-75-020
	Dog			30 mg/kg of feed, 90 d.	
	Cow	1700.			↓
	Birds				
	Bobwhite quail	3,300			
	Fish				
	Fathead minnow	13.3 mg/l 96 hr			
	Bluegill	49. mg/l 96 hr			
	Goldfish	31. mg/l 96 hr			
	Channel catfish	27. mg/l 96 hr			
	Rainbow trout	96. mg/l 96 hr			

Appendix Table I. (continued)

Chemical	Organism	Mammals and Birds Acute Oral LD_{50} mg/kg of Body Weight Aquatic Organisms Acute LC_{50} Mg/1	Chronic Effects	No Effect Level	Reference
Herbicides (cont.)					
Arsenical	Marine & Estuarine				
Monosodium	Scud			100. mg/1 96 hr	
Methane	Pink shrimp			1. mg/1 48 hr	
Arsonate	Eastern oyster			1. mg/1 48 hr	
Dalapon	Mammals				
Sodium salt	Rat	3860.			Ben-Dyke et al. 1970
	Fish				
	Fathead minnow	290. 96 hr			Surber & Pickering 1962
	Bluegill	290. 96 hr			Surber & Pickering 1962
	Coho salmon	340. 96 hr			Bond et al. 1960
	Crustaceans				
	Daphnia	11. 48 hr			Sanders & Cope 1966
	Insects				
	Stonefly			100 mg/1 96 hr	Sanders & Cope 1968

Appendix Table I. (continued)

Chemical	Organism	Mammals and Birds Acute Oral LD_{50} mg/kg of Body Weight Aquatic Organisms Acute LC_{50} Mg/l	Chronic Effects	No Effect Level	Reference
Herbicides (cont.)					
Dicamba	Mammals				
(Banvel)	Rat	1040 mg/kg			Bailey & White 1965
	Mouse	1190 mg/kg			
	Rabbit	2000 mg/kg			
	Guinea pig	3000 mg/kg			
	Birds				
	Chicken	673 mg/kg			Velsicol Chem. Corp. Bul. 521-2
	Pheasant	800 mg/kg			
	Fish				
	Bluegill	23 mg/l 96 hr			Velsicol Chem. Corp. Bul. 521-2
	Rainbow trout	28 mg/l 96 hr			
	Coho salmon Juv.	121 mg/l 48 hr			
	Carp	465 mg/l 48 hr			
	Crustaceans				
	Scud	3.9 mg/l 96 hr			Sanders 1969
	Daphnia			100. 48 hr	
	Crayfish			100. 48 hr	

Appendix Table I. (continued)

Chemical	Organism	Mammals and Birds Acute Oral LD_{50} mg/kg of Body Weight Aquatic Organisms Acute LC_{50} Mg/l	Chronic Effects	No Effect Level	Reference
Herbicides (cont.)					
	Mammals				
Picloram (Tordon)	Rat	8200.	> 500 mg/kg rat, oral	50 mg/kg rat (90 d)	Nat. Res. Council Canada Picloram, NRCC 13684
	Guinea pig	3000.		180 mg/kg sheep (90 d)	
				130 mg/kg calf (90 d)	
				150 mg/kg dog (2 yr)	
	Birds				
	Chicken	6,000			
	Bobwhite	10,000			
	Fish				
	Rainbow trout	34. mg/l			
	Coho salmon	29. mg/l			
	Bluegill	26.5 mg/l			
	Largemouth bass	19.7 mg/l			
	Fathead minnow	52. mg/l			

Appendix Table I. (continued)

Chemical	Organism	**Mammals and Birds** Acute Oral LD_{50} mg/kg of Body Weight Aquatic Organisms Acute LC_{50} Mg/l	Chronic Effects	No Effect Level	Reference
Herbicides (cont.)					
S-Triazines	**Mammals**				
Atrazine	Rat	1750. mg/kg			Bailey & White 1965
	Mouse	1750. mg/kg			Dalgaard-Mikkelsen & Paulsen 1962
	Sheep		500 mg/kg reduce wt. gain		
	Fish				
	Rainbow trout	4.5 mg/l 96 hr			Anon. 1971
	Bluegill	24. mg/l 96 hr			Anon. 1971
	Goldfish	60. mg/l 96 hr			Anon. 1971
	Channel catfish			10 mg/l 96 hr	Jones 1962
	Crustacean				
	Daphnia magna	3.6 mg/l 48 hr			Water Quality Criteria 1968
	Mammals				
Simazine	Rat	5000.			Bailey & White 1965
	Cow		750 mg/kg wt. loss		Palmer & Radeleff 1969
	Fish				
	Fathead minnow	6. mg/l 96 hr		.003 mg/l	
	Coho salmon	6.6 mg/l 48 hr			Bond *et al.* 1960

Appendix Table I. (continued)

Chemical	Organism	Mammals and Birds Acute Oral LD_{50} mg/kg of Body Weight Aquatic Organisms Acute LC_{50} Mg/1	Chronic Effects	No Effect Level	Reference
Herbicides (cont.)					
S-Triazines (cont.)	Fish (cont.)				
Simazine (cont.)	Bluegill	118. mg/l 48 hr			Cope 1964
	Rainbow trout	60. mg/l 48 hr			Cope 1964
	Crustacean				
	Crayfish			100. mg/l 48 hr	Sanders 1970
	Daphnia magna	1. mg/l 48 hr			Sanders 1970
Phenoxy	Mammals				
2,4,5-TP (Silvex)					
acid	Rat	650.			Rowe & Hymas 1954
butylester	Rat	600.		PGBE 30 mg/kg 90 days	Rowe & Hymas 1954
PGBE ester	Rat	620.			Rowe & Hymas 1954
PGBE ester	Guinea pig	1250.			Rowe & Hymas 1954
PGBE ester	Rabbit	819.	PGBE cow 25 mg/kg 20 doses PGBE sheep 25 mg/kg 10 doses		
	Birds				
PGBE ester	Chick	1190.			Rowe & Hymas 1954
acid	Mallard	2000.	33,700 mg/kg LD_{50}, 13 day dose House et al. 1967	500 mg/kg minor effects	Tucker & Crabtree 1970

Appendix Table I. (continued)

Chemical	Organism	Mammals and Birds Acute Oral LD_{50} mg/kg of Body Weight Aquatic Organisms Acute LC_{50} Mg/1	Chronic Effects	No Effect Level	Reference
Herbicides (cont.)					
Phenoxy	**Birds** (cont.)				
2,4,5-TP (Silvex)					
PGBE ester	Chicken	2000.	25,000 mg/kg 10 doses wt. loss Palmer & Radeleff 1969		Mullison 1966
	Fish				
PGBE ester	Bluegill	25. mg/1 48 hr			Hughes & Davis 1966
isooctyl ester	Bluegill	5. mg/1 48 hr			Hughes & Davis 1966
potassium salt	Bluegill	83. mg/1 48 hr			Hughes & Davis 1966
BE ester	Bluegill	2. mg/1 48 hr			Hughes & Davis 1966
TE amine	Bluegill	20. mg/1 48 hr			Hughes & Davis 1966
acid	Bluegill	2. mg/1 96 hr			Surber & Pickering 1961
acid	Fathead minnow	7.5 mg/1 96 hr			Surber & Pickering 1961
PGBE ester	Chinook	1.230 mg/1 48 hr			Bond 1959
PGBE ester	Largemouth bass	3.500 mg/1 24 hr			Bond 1959
	Crustaceans				
BE ester	Crayfish	60. mg/1 48 hr			Sanders 1971
PGBE ester	Crayfish			100. mg/1	Sanders 1971
	Marine & Estuarine				
	Eastern oyster, egg	.0059 mg/1 48 hr			Davis & Hidu 1969
	Eastern oyster, larvae	.710 mg/1 14 day			Davis & Hidu 1969

Appendix Table I. (continued)

Chemical	Organism	Mammals and Birds Acute Oral LD_{50} mg/kg of Body Weight Aquatic Organisms Acute LC_{50} Mg/l	Chronic Effects	No Effect Level	Reference
Herbicides (cont.)					
Phenoxy (cont.)	Mammals				
2,4-D acid	Rat	375	TEA 1000 mg/l in water (120 days) slower growth Bjorklund & Erne 1966	30 mg/kg (4 weeks)	Rowe & Hymas 1954
	Mouse	368			
	Birds				
	Chicken	540	TEA 1000 mg/l in water, fewer eggs. Bjorklund & Erne 1966		Tucker & Crabtree 1970
	Mallard	2000			
2,4-D ester	Insect				
BE ester	Stonefly	1.6 mg/l 96 hr			Sanders & Cope 1968
	Crustacean				
	Crayfish	60. mg/l 48 hr			Sanders 1970
2,4-D salt					
DE amine	Crayfish			100. mg/l 48 hr	Sanders 1970

Appendix Table I. (continued)

Chemical	Organism	Mammals and Birds Acute Oral LD_{50} mg/kg of Body Weight Aquatic Organisms Acute LC_{50} Mg/l	Chronic Effects	No Effect Level	Reference
Herbicides (cont.)					
Phenoxy (cont.)	Fish				
2,4-D acid	Bluegill	390. mg/l	5. mg/l temporary liver changes. Cope *et al.* 1970	1. mg/l	Davis & Hardcastle 1959
	Bass	375. mg/l			Davis & Hardcastle 1959
2,4-D					
DE amine	Bluegill	166. mg/l 48 hr			Lawrence 1964
alkaloamine	Bluegill	435. mg/l 48 hr			Lawrence 1964
2,4-D ester					
BE ester	Fathead minnow	5.6 mg/l 96 hr		.3 mg/l 10 mos.	Mount & Stephan 1967
PGBE ester	Bluegill	3. mg/l 48 hr			
isooctyl ester	Bluegill	9. mg/l 48 hr			Lawrence 1964
butyl ester	Bluegill	1. mg/l 48 hr			Lawrence 1964
isopropyl ester	Bluegill	1. mg/l 48 hr			Lawrence 1964

Appendix Table I. (continued)

Chemical	Organism	Mammals and Birds Acute Oral LD_{50} mg/kg of Body Weight Aquatic Organisms Acute LC_{50} Mg/1	Chronic Effects	No Effect Level	Reference
Herbicides (cont.)					
Phenoxy					
2,4,5-T	Mammals		No Effect Level		
acid	Rat	500.	PGBE cattle 100 mg/kg/d, 10 d	Palmer & Radeleff 1969	Rowe & Hymas 1954
amyl ester	Rat	750.	PGBE sheep 50 mg/kg/d, 10 d		Rowe & Hymas 1954
isopropyl ester	Rat	495.	TEA sheep 100 mg/kg/d, 481 d		Rowe & Hymas 1954
acid	Mouse	389.			Rowe & Hymas 1954
isopropyl ester	Mouse	550.			Rowe & Hymas 1954
butyl ester	Mouse	940.			Rowe & Hymas 1954
acid	Guinea pig	380.	Chronic Effects		Rowe & Hymas 1954
acid	Dog	100.	10 mg/kg/d, 90 d, minor wt. loss Drill & Hiratzka 1953		Dalgaard-Mikkelsen & Paulsen 1962
	Birds				
acid	Chicken	310.		PGEE 10 mg/kg/d 10 days Palmer & Radeleff 1969	Rowe & Hymas 1954
	Fish				
BE ester	Bluegill	1.4 mg/l 48 hr			Hughes & Davis 1963
isooctyl ester	Bluegill	10.-31. mg/l 48 hr			Hughes & Davis 1963
PGBE ester	Bluegill	17. mg/l 48 hr			Hughes & Davis 1963
BE ester	Bluegill	1.4 mg/l 48 hr			Bond et al. 1959
dimethyl amine	Bluegill	144. mg/l 48 hr			Bond et al. 1959

Appendix Table I. (continued)

Chemical	Organism	Mammals and Birds Acute Oral LD_{50} mg/kg of Body Weight Aquatic Organisms Acute LC_{50} Mg/l	Chronic Effects	No Effect Level	Reference
Herbicides (cont.)					
Phenoxy (contaminant)					
2,3,7,8-tetrachlorodibenzoidioxin (impurity in silvex and 2,4,5-T) (TCDD)					
	Mammals				
	Rat	.022			Schwetz et al. 1973
	Mouse	.114			Vos et al. 1973
	Rabbit	.115			Rosen & Kraybill 1966
	Dog	3.			Rosen & Kraybill 1966
	Guinea pig	.0006			Rosen & Kraybill 1966
	Fish				
	Rainbow trout	.0000054 mg/l 120 hrs			Miller et al. 1973
	Coho salmon	< .000023 mg/l 120 hrs			Miller et al. 1973
	Guppy	< .0001 mg/l			Norris & Miller 1974

Appendix Table I. (continued)

Chemical	Organism	Mammals and Birds Acute Oral LD_{50} mg/kg of Body Weight Aquatic Organisms Acute LC_{50} Mg/l	Chronic Effects	No Effect Level	Reference
Insecticides					
Carbamates	Mammals				
Carbaryl (Sevin)	Rat	500.			Matsumura 1975
	Guinea pig	280.			Pest. Chem. Of. Comp. p. 192, 1966
	Rabbit	710.			Pest. Chem. Of. Comp. p. 192, 1966
	Birds				
	Chicken	197.			Sherman & Ross 1961
	Fish				
	Fathead minnow	9.000 mg/l 96 hr		Fathead minnow .0002	Carlson NWQL
	Bluegill	6.760 mg/l 96 hr			Macek & McAllister 1970
	Largemouth bass	6.400 mg/l 96 hr			Macek & McAllister 1970
	Rainbow trout	1.950 mg/l 96 hr			Macek & McAllister 1970
	Brown trout	1.950 mg/l 96 hr			Macek & McAllister 1970
	Coho salmon	.764 mg/l 96 hr			Macek & McAllister 1970
	Channel catfish	15.800 mg/l 96 hr			Macek & McAllister 1970
	Black bullhead	20.000 mg/l 96 hr			Macek & McAllister 1970
	Carp	5.280 mg/l 96 hr			Macek & McAllister 1970
	Gold fish	13.200 mg/l 96 hr			Macek & McAllister 1970
	Perch	.745 mg/l 96 hr			Macek & McAllister 1970
	Insects				
	Stonefly	.0048			Sanders & Cope 1968

Appendix Table I. (continued)

Chemical	Organism	Mammals and Birds Acute Oral LD_{50} mg/kg of Body Weight Aquatic Organisms Acute LC_{50} Mg/1	Chronic Effects	No Effect Level	Reference
Insecticides (cont.)					
Carbamates (cont.)	Crustaceans				
Carbaryl (Sevin) (cont.)	Crayfish	.0086 mg/1 96 hr			Sanders & Cope 1968
	Red crayfish	2.000 mg/1 72 hr			Muncy & Oliver 1963
	Daphnia pulex	.0064 mg/1 48 hr			Sanders & Cope 1966
	Marine & Estuarine				
	Dungeness crab	.180 mg/1 96 hr			Buchanan et al. 1967
	Am. oyster larv.	3.000 mg/1 14 day			Davis & Hidu 1969
	Pac. oyster larv.	2.200 mg/1 96 hr			Stewart et al. 1967
	Bay mussel larv.	2.300 mg/1 96 hr			Stewart et al. 1967
	Mud shrimp	.090 mg/1 48 hr			Stewart et al. 1967
	Ghost shrimp larv.	.080 mg/1 48 hr			Stewart et al. 1967
	Bent-nosed clam	1.700 mg/1 96 hr			Armstrong & Millemann 1974
	Cockle clam	3.750 mg/1 96 hr			Butler et al. 1968
Organochlorine	Mammals				
Endosulfan (Thiodan)	Rat	100.			Schafer 1972
		30.-79.			Matsumura 1975
	Birds				
	Duck	34.			Matsumura 1972
	Cowbird	1200.			DeWitt et al. 1962
	Bobwhite, young	380.			DeWitt et al. 1962
	Pheasant	850			DeWitt et al. 1962

Appendix Table I. (continued)

Chemical	Organism	Mammals and Birds Acute Oral LD_{50} mg/kg of Body Weight Aquatic Organisms Acute LC_{50} Mg/l	Chronic Effects	No Effect Level	Reference
Insecticides (cont.)					
Organochlorine (cont.)	Fish				
Endosulfan (Thiodan) (cont.)	Rainbow trout	.0003 mg/l 96 hr			Schoettger 1970
	White sucker	.003 mg/l 96 hr			Schoettger 1970
	Insects				
	Stonefly	.0023 mg/l 96 hr			Sanders & Cope 1968
	Damselfly naiads	.072 mg/l 96 hr			Schoettger 1970
	Crustaceans				
	Daphnia magna	.052 mg/l 96 hr			Schoettger 1970
	Scud	.006 mg/l 96 hr			Sanders 1972
	Marine & Estuarine				
	Bay mussel		1000 mg/l, delayed spawning		Lin & Lee 1975
	Mammals				
Endrin	Mouse	1.37			
	Rat	3.			
	Rabbit	7.			
	Guinea pig	16.		Dog .02 mg/kg/day	Treon et al. 1955
	Birds				
	Pigeon	5.6			

Appendix Table I. (continued)

Chemical	Organism	Mammals and Birds Acute Oral LD_{50} mg/kg of Body Weight Aquatic Organisms Acute LC_{50} Mg/l	Chronic Effects	No Effect Level	Reference
Insecticides (cont.)					
Organochlorine (cont.)	Fish				
Endrin (cont.)	Fathead minnow	.0005 mg/l 96 hr			Henderson et al. 1959
	Bluegill	.0006 mg/l 96 hr			Henderson *et al.* 1959
	Rainbow trout	.0006 mg/l 96 hr			Katz 1961
	Coho salmon	.0005 mg/l 96 hr			Katz 1961
	Chinook salmon	.0012 mg/l 96 hr			Katz 1961
	Cutthroat trout	.00011 mg/l 96 hr			Post & Schroeder 1971
	Striped bass (juv.)	.000094 mg/l 96 hr			Korn & Earnest 1974
	Insects				
	Stonefly	.0024 mg/l 96 hr			Jensen & Gaufin 1966
	Stonefly	.00032 mg/l 96 hr			Jensen & Gaufin 1966
	Crustaceans				
	Scud	.0009 mg/l 120 hr			Sanders 1972
	Crayfish	.0032 mg/l 96 hr			Sanders 1972
	Daphnia	.020 mg/l 48 hr			Sanders & Cope 1966
	Marine & Estuarine				
	Sand shrimp	.0017 mg/l 96 hr			Eisler 1969
	Hermit crab	.012 mg/l 96 hr			Eisler 1969

Appendix Table I. (continued)

Chemical	Organism	Mammals and Birds Acute Oral LD_{50} mg/kg of Body Weight Aquatic Organisms Acute LC_{50} Mg/1	Chronic Effects	No Effect Level	Reference
Insecticides (cont.)					
Organochlorine	Mammals				
Lindane	Rat	88.			Gaines 1960
	Rabbit	60.			Woodard & Hagan 1947
	Guinea pig	127.			Woodard & Hagan 1947
	Birds				
	Chicken	250.			Grolleau 1965
	Pheasant	75.-100.			Grolleau 1965
	Chukar	35.-185.			Grolleau 1965
	Bobwhite	120.-130.			Dahlen & Gaugen 1954
	Morning dove	350.-400.			Dahlen & Gaugen 1954
	Fish				
	Fathead minnow	.087 mg/1 96 hr			Macek & McAllister 1970
	Bluegill	.068 mg/1 96 hr			Macek & McAllister 1970
	Redear sunfish	.083 mg/1 96 hr			Macek & McAllister 1970
	Largemouth bass	.032 mg/1 96 hr			Macek & McAllister 1970
	Rainbow trout	.027 mg/1 96 hr			Macek & McAllister 1970
	Brown trout	.002 mg/1 96 hr			Macek & McAllister 1970
	Coho salmon	.041 mg/1 96 hr			Macek & McAllister 1970
	Perch	.068 mg/1 96 hr			Macek & McAllister 1970

Appendix Table I. (continued)

Chemical	Organism	Mammals and Birds Acute Oral LD_{50} mg/kg of Body Weight Aquatic Organisms Acute LC_{50} Mg/l	Chronic Effects	No Effect Level	Reference
Insecticides (cont.)					
Organochlorine (cont.)	Fish (cont.)				
Lindane	Channel catfish	.044 mg/l 96 hr			Macek & McAllister 1970
	Guppies	.138 mg/l 96 hr			Tarzwell 1958
	Gold fish	.152 mg/l 96 hr			Tarzwell 1958
	Carp	.090 mg/l 96 hr			Macek & McAllister 1970
	Black bullhead	.064 mg/l 96 hr			Macek & McAllister 1970
	Insects				
	Stonefly	.0045 mg/l 96 hr			Sanders & Cope 1968
	Crustaceans				
	Daphnia pulex	.460 mg/l 96 hr			Sanders & Cope 1966
	Marine & Estuarine				
	Eastern oyster - egg	9.100 mg/l 48 hr			Davis & Hidu 1969
	Hard clam - larvae	> 10.000 mg/l 12 day			Davis & Hidu 1969
	Hermit crab	.005 mg/l 96 hr			Eisler 1969
Organophosphorus	Mammals				
Diazinon	Rat	76.			Schafer 1972
	Mouse	85.			Guide to Chem. Used in Crop Prot. 6:171, 1973

Appendix Table I. (continued)

Chemical	Organism	Mammals and Birds Acute Oral LD_{50} mg/kg of Body Weight Aquatic Organisms Acute LC_{50} Mg/1	Chronic Effects	No Effect Level	Reference
Insecticides (cont.)					
Organophosphorus (cont.)	Birds				
Diazinon (cont.)	Bird				Schafer 1972
	Mallard	3.5			Tucker & Crabtree 1970
	Pheasant	4.3			Tucker & Crabtree 1970
	Fish				
	Rainbow trout	170. mg/l 48 hr			Cope 1964
	Bluegill	30. mg/l 48 hr			Cope 1964
	Insects				
	Stonefly	25. mg/l 96 hr			Sanders & Cope 1968
	Crustaceans				
	Daphnia pulex	.9 mg/l 48 hr			Sanders & Cope 1966
	Daphnia magna			.26 mg/l	Biesinger NWQL
	Mammals				
Disulfoton	Rat	12.5			Frawley et al. 1963
	Rat - skin	6.			Gaines 1969
	Birds				
	Bobwhite, young	800.			DeWitt et al. 1962

Appendix Table I. (continued)

Chemical	Organism	Mammals and Birds Acute Oral LD_{50} mg/kg of Body Weight Aquatic Organisms Acute LC_{50} Mg/1	Chronic Effects	No Effect Level	Reference
Insecticides (cont.)					
Organophosphorus (cont.)	Fish				
Disulfoton (cont.)	Fathead minnow	.063 mg/1 96 hr			Pickering et al. 1962
	Bluegill	3.700 mg/1 96 hr			Pickering et al. 1962
	Insects				
	Stonefly	.005	.0017 mg/1 (30 day LC_{50}) Jensen & Gaufin 1964		Sanders & Cope 1968
	Crustaceans				
	Scud	.027			Sanders 1972
	Glass shrimp	.038			Sanders 1972
	Mammals				
Fenitrothion (Sumithion)	Rat	250.			Schafer 1972
	Mouse	715.			Cherkinskii et al. 1966
	Birds				
	Chicken	280.			Sherman et al. 1967
	Wild bird	25.			Schafer 1972
	Fish				
	Brook trout		Loss of established hierarchical order. 4 weeks of 10 mg/g dosage in food.		Wildish & Lister 1973

Appendix Table I. (continued)

Chemical	Organism	Mammals and Birds Acute Oral LD_{50} mg/kg of Body Weight Aquatic Organisms Acute LC_{50} Mg/l	Chronic Effects	No Effect Level	Reference
Insecticides (cont.)					
Organophosphorus (cont.)	Fish (cont.)				
Fenitrothion (cont.) (Sumithion)	Atlantic salmon	1.000 mg/l 96 hr	1. mg/l, 16 hrs loss of territorrial behavior. 24 hrs loss of learning ability (Symons 1973), and increasing predation by brook trout (Hatfield et al. 1972).		Hatfield & Johansen 1972
	Mammals				
Guthion azinphosmethyl	Rat	15.			Chem. of Pest. 1971
	Guinea pig	80.			Pest. Chem. Office Corp., p. 570, 1966
	Birds				
	Chicken	277.			Sherman et al. 1967
	Fish				
	Rainbow trout	.014 mg/l 96 hr			Macek & McAllister 1970
	Brown trout	.017 mg/l 96 hr			Macek & McAllister 1970
	Coho salmon	.005 mg/l 48 hr			Katz 1961
	Chinook salmon	.0062 mg/l 48 hr			Katz 1961
	Redear sunfish	.052 mg/l 96 hr			Macek & McAllister 1970
	Bluegill	.022 mg/l 96 hr			Macek & McAllister 1970
	Fathead minnow	.093 mg/l 96 hr			Henderson 1959

Appendix Table I. (continued)

Chemical	Organism	Mammals and Birds Acute Oral LD_{50} mg/kg of Body Weight Aquatic Organisms Acute LC_{50} Mg/1	Chronic Effects	No Effect Level	Reference
Insecticides (cont.)					
Organophosphorus (cont.)	Fish (cont.)				
Guthion azinphosmethyl (cont.)	Channel catfish	3.290 mg/1 96 hr			Macek & McAllister 1970
	Black bullhead	3.500 mg/1 96 hr			Macek & McAllister 1970
	Goldfish	4.270 mg/1 96 hr			Macek & McAllister 1970
	Insects				
	Stonefly	.0015 mg/1 96 hr			Sanders & Cope 1968
	Marine & Estuarine				
	Eastern oyster, egg	.620 mg/1 48 hr			Davis & Hidu 1969
	Hard clam, larvae	.860 mg/1 12 day			Davis & Hidu 1969
	Mammals				
Malathion	Rat	599.			Boyd & Taylor 1971
	Fish				
	Fathead minnow	9.000 mg/1 96 hr	.580 10 mo., spinal defects		Mount & Stephan 1967
	Bluegill	.110 mg/1 96 hr	.0074 spinal defects	.200 10 mo.	Eaton 1971
	Redear sunfish	.170 mg/1 96 hr		.0036 11 mo.	Macek & McAllister 1970

Appendix Table I. (continued)

Chemical	Organism	Mammals and Birds Acute Oral LD_{50} mg/kg of Body Weight Aquatic Organisms Acute LC_{50} Mg/1	Chronic Effects	No Effect Level	Reference
Insecticides (cont.)					
Organophosphorus (cont.)	Fish (cont.)				
Malathion (cont.)	Largemouth bass	.285 mg/1 96 hr			Macek & McAllister 1970
	Rainbow trout	.170 mg/1 96 hr	.112 10 days Post et al. 1974		Macek & McAllister 1970
	Brown trout	.200 mg/1 96 hr	45% reduction in AChE		Macek & McAllister 1970
	Coho salmon	.100 mg/1 96 hr			Macek & McAllister 1970
	Perch	.263 mg/1 96 hr			Macek & McAllister 1970
	Channel catfish	8.970 mg/1 96 hr			Macek & McAllister 1970
	Carp	6.590 mg/1 96 hr			Macek & McAllister 1970
	Goldfish	10.700 mg/1 96 hr			Macek & McAllister 1970
	Black bullhead	12.900 mg/1 96 hr			Macek & McAllister 1970
	Insects				
	Stonefly	.010 mg/1 96 hr			Sanders & Cope 1968
	Crustaceans				
	Red crayfish			20 mg/1	Muncy & Oliver 1964
	Crayfish	.180 mg/1 96 hr			Sanders 1970
	Daphnia pulex	.0018 mg/1 96 hr			Sanders & Cope 1966

Appendix Table I. (continued)

Chemical	Organism	Mammals and Birds Acute Oral LD_{50} mg/kg of Body Weight Aquatic Organisms Acute LC_{50} Mg/1	Chronic Effects	No Effect Level	Reference
Insecticides (cont.)					
Organophosphorus (cont.)	**Marine & Estuarine**				
Malathion (cont.)	Eastern oyster egg	9.070 mg/1 48 hr			Davis & Hidu 1969
	Eastern oyster larvae	2.660 mg/1 12 day			Davis & Hidu 1969
	Bay mussel embryo	13.400 mg/1 12 day			Liu & Lee 1975
	Hermit crab	.083 mg/1 96 hr			Eisler 1969
	Mammals				
Phosphamidon (Dimecron)	Rat	17.			Schafer 1972
	Mouse	10.			Sachsse & Voss 1971
	Rabbit	70.			Sachsse & Voss 1971
	Dog	50.			
	Birds				
	Mallard	3.			Sachsse & Voss 1971
	Pigeon	3.			Sachsse & Voss 1971
	Chukar	9.			Sachsse & Voss 1971
	Fish				
	Fathead minnow	100. mg/1 96 hr			FPRL
	Bluegill	4.5 mg/1 96 hr			FPRL
	Channel catfish	70. mg/1 96 hr			FPRL

Appendix Table I. (continued)

Chemical	Organism	Mammals and Birds Acute Oral LD_{50} mg/kg of Body Weight Aquatic Organisms Acute LC_{50} Mg/l	Chronic Effects	No Effect Level	Reference
Insecticides (cont.)					
Organophosphorus (cont.)	Insects				
Phosphamidon (Dimecron)	Stonefly	.15 mg/l 96 hr			Sanders & Cope 1968
	Crustaceans				
	Scud	.016 mg/l 96 hr			Sanders 1972
	Crayfish	7.5 mg/l 96 hr			Sanders 1972
	Daphnia	.0088 mg/l 96 hr			Sanders & Cope 1968
	Mammals				
Trichlorfon (Dipterex)	Mouse	600.			Schafer 1972
	Rat	400.			
	Fish				
	Fathead minnow	109. mg/l 96 hr			Pickering *et al.* 1962
	Bluegill	3.8 mg/l 96 hr			Pickering *et al.* 1962
	Insects				
	Stonefly	.069 mg/l 96 hr			Jensen & Gauffin 1966
	Stonefly	.0165 mg/l 96 hr			Jensen & Gauffin 1966
	Crustaceans				
	Scud	.040 mg/l 96 hr			Sanders & Cope 1966
	Daphnia	.00018 mg/l 96 hr			Sanders & Cope 1966

Appendix Table I. (continued)

Chemical	Organism	Mammals and Birds Acute Oral LD_{50} mg/kg of Body Weight Aquatic Organisms Acute LC_{50} Mg/l	Chronic Effects	No Effect Level	Reference

Abreviations: AChE - acetylcholinesterase
BE - butoxy ethyl
PGBE - propyleneglyco butylether
TE - triethyl
TEA - triethylamine
DE - diethyl
d - day or days

Appendix II

REFERENCES FOR APPENDIX TABLE 1

Armstrong, D. A., and R. E. Millemann. 1974. Pathology of acute poisoning with the insecticide Sevin in the bent-nosed clam, Macoma nasuta. J. Invert. Path. 24:201-212.

Bailey, George W., and Joe L. White. 1965. Herbicides: a compilation of their physical, chemical, and biological properties. Res. Rev. 10:97-122.

Ben-Dyke, R., D. M. Sanderson, and Diana N. Noakes. 1970. Acute toxicity data for pesticides. World Review of Pest Control 9:119-127.

Biesinger, K. E. 1971. Unpublished data, Env. Prot. Agency, Nat. Water Quality Lab., Duluth, Minnesota 55804.

Bond, C. E., R. H. Lewis, and J. L. Fryer. 1960. Toxicity of various herbicidal materials to fishes. Robert A. Taft Sanit. Eng. Center Tech. Rept. W60-3:96-101.

Bough, R. G., E. E. Cliffe, and B. Lessel. 1965. Comparative toxicity and blood level studies on Binapacryl and DNBP. Toxicology and Applied Pharmacology 7:353-360.

Boyd, Eldon M., and Frances I. Taylor. 1971. Toxaphene toxicity in protein-deficient rats. Toxicology and Applied Pharmacology 18:158-167.

Buchanan, D. V., R. E. Millemann, and N. E. Stewart. 1970. Effects of the insecticide Sevin on various stages of the Dungeness crab, Cancer magister. J. Fish. Res. Bd. Can. 27:93-104.

Butler, J. A., R. E. Millemann, and N. E. Stewart. 1968. Effects of the insecticide Sevin on survival and growth of the cockle clam Clinocardium nuttallii. J. Fish Res. Bd. Can. 25:1621-1635.

Cherkinskii, S. N., K. I. Akulov, and G. N. Krasovskii. 1966. Relative toxicity of various pesticides which may pollute water sources. Hygiene and Sanitation 31:18-22.

Cope, Oliver B. 1964. Sport fishery investigations. In: The effects of pesticides on fish and wildlife. USDI FW Service Circ. 226. p. 51-63.

Dalgaard-Mikkelsen, S. and E. Poulsen. 1962. Toxicology of herbicides. Pharmacological Revues 14:225.

Davis, H. C., and H. Hidu. 1969. Effects of pesticides on embryonic development of clams and oysters and on survival and growth of the larvae. U.S. Fish and Wildl. Serv., Fish. Bull. 67:393-404.

DeWitt, J. B., W. H. Stickel, and P. F. Springer. 1963. Wildlife studies. Patuxent Wildl. Res. Cntr. U.S. Dept. Int. Fish and Wildl. Serv. Circ. 167:74-96.

Drill, V. A., and T. Hiratzka. 1953. Toxicity of 2,4-dichlorophenoxyacetic acid and 2,4,5-trichlorophenoxyacetic acid, a report on their acute and chronic toxicity in dogs. Arch. Ind. Hyg. Occupt. Med. 7:61-67.

Eaton, J. G. 1970. Chronic malathion toxicity to the bluegill (Lepomis macrochirus Rafinesque). Water Res. 4:673.

Eisler, R. 1969. Acute toxicities of insecticides to marine decapod crustaceans. Crustaceana 16(3):302-310.

Frawley, J. P., R. Weir, T. Tusing, K. P. DuBois, and J. C. Calandra. 1963. Toxicological investigations on Delnav. Toxicology and Applied Pharmacology 5:605-624.

Gaines, Thomas B. 1960. The acute toxicity of pesticides to rats. Toxicology and Applied Pharmacology 2:88-99.

Gaines, Thomas B. 1969. Acute toxicity of pesticides. Toxicology and Applied Pharmacology 14:515-534.

Hatfield, C. T., and P. H. Johnsen. 1972. Effects of four insecticides on the ability of Atlantic salmon (Salmo salar) parr to learn and retain a simple conditioned response. J. of Fisheries Res. Bd. Can. 29:315-321.

Henderson, Crosswell, and Quentin H. Pickering. 1958. Toxicity of organic phosphorus insecticides to fish. Trans. of the American Fisheries Soc. 87:39-51.

Henderson, C., Q. H. Pickering, and C. M. Tarzwell. 1959. Relative toxicity of 10 chlorinated hydrocarbon insecticides to four species of fish. Trans. Am. Fish. Soc. 88:23-32.

House, W. B. et al. 1967. Assessment of ecological effects of extensive or repeated use of herbicides. Final report of Midwest Research Institute, Project 3103-B.

Hughes, J. S., and J. T. Davis. 1962. Toxicity of selected herbicides to bluegill sunfish. La. Acad. Sci. Proc. 25:86-93.

Hughes, Janis S., and James T. Davis. 1963. Variations in toxicity to bluegill sunfish of phenoxy herbicides. Weeds 11:50-53.

Hughes, J. S., and J. T. Davis. 1964. Effects of selected herbicides on bluegill sunfish. Proc. Southeast Ass. Game Fish Comm. 18:480-482.

Inglis, Anthony, and Edward L. Davis. 1972. Effects of water hardness of several organic and inorganic herbicides to fish. Bur. Sp. Fish and Wildl. Tech. Bull. 66. 22 p.

Jensen, L. D., and A. R. Gaufin. 1966. Acute and long-term effects of organic insecticides on two species of stonefly naiads. J. Wat. Pollut. Cont. Fed. 38:1273-1286.

Jones, R. O. 1962. Tolerance of fry to some common warm water fishes to some chemicals employed in fish culture. Southeast Assoc. Game and Fish Commrs. Conf. Proc. 16:436-445.

Katz, M. 1961. Acute toxicity of some organic insecticides to three species of salmonids and to the three spine stickleback. Trans. of the Amer. Fisheries Soc. 90:264.

Knepp, G. L., and G. F. Arkin. 1973. Ammonia toxicity levels and nitrate tolerance of channel catfish. The progressive Fish-Culturalist 35(4): 221-224.

Korn, Sid, and Russell Earnest. 1974. Acute toxicity of twenty insecticides to striped bass, *Morone saxatillis*. Calif. Fish and Game 60(3):128-

Lin, David H. W., and Jean M. Lee. 1975. Toxicity of selected pesticides to the bay mussel (*Mytilus edulis*). EPA 660/3-75-016.

Macek, K. J., and W. A. McAllister. 1970. Insecticide susceptibility of some common fish family representatives. Trans. of Am. Fisheries Soc. 99:20-27.

Miller, R. A., Logan Norris, and C. L. Hawkes. 1973. Toxicity of 2,3,7,8-TCDD in aquatic organisms. Envir. Health Persp. 5:177-86.

Mount, D. I., and C. E. Stephan. 1967. A method for estimating acceptable toxicant limits for fish - Malathion and Butoxyethanol ester of 2,4-D. Tran. Am. Fish. Soc. 96:185-193.

Mullison, W. R. 1966. Some toxicological aspects of silvex. Dow Chem. Co. 16 p.

Muncy, R. J., and A. D. Oliver. 1963. Toxicity of ten insecticides to the red crayfish, *Procambarus clarki* (Girard). Trans. Am. Fish. Soc. 92: 428-431.

National Research Council of Canada. 1974. Picloram: the effects of its use as a herbicide on environmental quality. Associate Committee on Scientific Criteria for Environmental Quality. 127 p.

Norris, L. A., and R. A. Miller. 1974. The toxicity of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) in guppies (Poecilia reticulatus Peters). Bul. Env. Contam. and Toxicol. 12:76-80.

Palmer, J. S., and R. D. Radeleff. 1969. The toxicity of some organic herbicides to cattle, sheep and chickens. USDA Agr. Res. Serv., Wash., D.C. 26 p.

Pickering, Q. H., C. Henderson, and A. E. Lemke. 1962. The toxicity of organic phosphate insecticides to different species of warm water fishes. Trans. of Am. Fish. Soc. 91:175-184.

Post, G., and T. R. Schroeder. 1971. The toxicity of four insecticides to four salmonid species. Bulletin of Environmental Contamination 6:144.

Rosen, A. A., and H. F. Kraybill, eds. 1966. Organic pesticides in the envionment. Advances in Chemistry Series 60. American Chemical Society, Wash., D.C.

Rowe, V. K., and T. A. Hymas. 1954. Summary of toxicological information on 2,4-D and 2,4,5-T type herbicides and an evaluation of the hazards to livestock associated with their use. Amer. J. Veterinary Research 15:622-629.

Russo, R. C., C. E. Smith, and R. V. Thurston. 1974. Acute toxicity of nitrite to rainbow trout (Salmo gairdneri). J. of Fisheries Res. Bd. Canada 31:1653-1655.

Sanders, Herman O. 1970. Toxicities of some herbicides to six species of freshwater crustaceans. J. Wat. Pollut. Contr. Fed. 42:1544-1550.

Sanders, Herman O. 1972. Toxicity of some insecticides to four species of malacostracan crustaceans. Bur. Sp. Fish. and Wildl. Tech. Bul. 66. 19 p.

Sanders, H. W., and Oliver B. Cope. 1966. Toxicities of several pesticides to two species of cladocerans. Trans. Am. Fish Soc. 95:165-169.

Sanders, H. O., and O. B. Cope. 1968. Toxicity of several pesticides to naiads of three species of stoneflies. Limnology and Oceanography 13: 112-117.

Schafer, Edward W. 1972. The acute oral toxicity of 369 pesticidal, pharmaceutical and other chemicals to wild birds. Toxicology and Applied Pharmacology 21:315-330.

Schoettger, R. A. 1970. Toxicology of thiodan in several fish and aquatic invertebrates. Bur. Sp. Fish. and Wildl. Inv. in Fish Control #35. 31 p. USG Printing Office.

Schwetz, B. A., J. M. Norris, G. L. Sparschu, V. K. Rowe, P. J. Gehring, J. L. Emerson, and C. B. Gerbig. 1973. Toxicology of chlorinated dibenzo-p-dioxins. Envir. Health Perspectives 5:87-99.

Sherman, Martin, and Ernest Ross. 1961. Acute and subacute toxicity of insecticides to chicks. Toxicology and Applied Pharmacology 3:521-533.

Sherman, Martin, R. B. Herrick, Ernest Ross, and M. T. Y. Chang. 1967. Further studies on the acute and subacute toxicity of insecticides to chicks. Toxicology and Applied Pharmacology 11:49-67.

Smith, C. E., and W. G. Williams. 1974. Experimental nitrite toxicity in rainbow trout and chinook salmon. Transactions of the American Fisheries Society 103:389.

Smith, R. Blackwell, Jr., P. S. Larson, J. K. Finnegan, H. B. Haag, Gordon H. Hennigar, and Florapearl Cobey. 1959. Toxicologic studies on 2,2-bis-(chlorophenyl)-2,2,2-trichloroethanol (Kelthane). Toxicology and Applied Pharmacology 1:119-134.

Stewart, N. E., R. E. Milleman, and W. P. Breese. 1967. Acute toxicity of the insecticide Sevin and its hydrolytic product 1-naphthol to some marine organisms. Trans. Am. Fish. Soc. 96:25-30.

Surber, E. W., and Q. H. Pickering. 1962. Acute toxicity of endothal, diquat, Hyamine, dalapon, and silvex to fish. Progr. Fish Cult. 24: 164-171.

Tarzwell, C. M. 1958. The toxicity of some organic insecticides to fishes. Proc. Ann. Conf. Seast. Assoc. Game and Fish Comm. 12:233-239.

Trama, F. B. 1954. The acute toxicity of some common salts of sodium, potassium, and calcium to the common bluegill (L. macrochirus Rafinesque). Proc. Acad. Nat. Sci. Phila. 106:185-214.

Treon, J. F., F. P. Cleveland, and J. Cappel. 1955. Toxicity of endrin for laboratory animals. Jour. Agric. and Food Chemistry 3:842-848.

Tucker, R. K., and D. Glen Crabtree. 1970. Handbook of toxicity of pesticides to wildlife. USDI, Bur. S. Fish. and Wildl. Resource Pub. #84.

U.S. Environmental Protection Agency. 1975. Initial scientific review of cacodylic acid. EPA 540/1-75-021.

U.S. Environmental Protection Agency. 1975. Initial scientific review of MSMA/DSMA. 113 p. EPA 540/1-75-020.

Vos, J. G., J. A. Moore, and J. G. Zinkl. 1973. Toxicity of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) in C57Bl/6 mice. Toxicology and Applied Pharmacology 29:229-241.

Water Quality Criteria. 1968. Report of the National Technical Adv. Committee to the Secretary of Interior. Fed. Water Pollution Control Adm., U.S. Dept. of Int.

Westin, Deborah T. 1974. Nitrate and nitrite toxicity to salmonoid fishes. Prog. Fish Cult. 36(2):86-89.

Wildish, D. J., and N. A. Lister. 1973. Biological effects of fenitrothion in the diet of brook trout. Bull. of Env. Cont. and Tox. 10:333-339.

Woodard, G., and E. C. Hagan. 1947. Toxicity studies on the isomers and mixtures of isomers of benzene hexachloride. Federation Proceedings 6:386.

Appendix III

CHECKLIST OF SPECIES FOR WHICH TOXICITY DATA IS GIVEN

Fish

Common Name	Scientific Name
Atlantic salmon	*Salmo salar*
Brook trout	*Salvelinus fontinalis*
Brown trout	*Salmo trutta*
Brown bullhead	*Ictalurus nebulosus*
Black bullhead	*Ictalurus melas*
Bluegill sunfish	*Lepomis machrochirus*
Channel catfish	*Ictalurus punctatus*
Carp	*Cyprinus carpio*
Chinook salmon	*Oncorhynchus tschawytscha*
Coho salmon	*Oncorhynchus kisutch*
Cutthroat trout	*Salmo clarki*
Fathead minnow	*Pimephales promelas*
Green sunfish	*Lepomis cyanellus*
Goldfish	*Carassius auratus*
Guppy	*Lebistes reticulatus*
Largemouth bass	*Micropterus salmoides*
Perch	*Perca flavescens*
Rainbow trout	*Salmo gairdnerii*
Redear sunfish	*Lepomis microlophus*
Striped bass	*Morone saxatilis*
White sucker	*Catostomus commersoni*

Crustaceans

Common Name	Scientific Name
Crayfish	*Oronectes nais*
Crayfish, red	*Procambarus clarki*
Dungeness crab	*Cancer magister*
Hermit crab	*Pagurus longicarpus*
Ghost shrimp	*Callianassa californiensis*
Glass shrimp	*Palaemonetes kadiakensis*
Mud shrimp	*Upogebia pugettensis*
Sand shrimp	*Crangon septemspinosa*
Scud	*Gammarus fasciatus*
Water flea	*Daphnia magna*
Water flea	*Daphnia pulex*

Molluscs

Bay mussel	*Mytillus edulis*
Bent-nosed clam	*Macoma nasuta*
Cockle clam	*Clinocardium nuttallii*
Eastern oyster	*Crassostrea virginica*
Hard clam	*Mercenaria mercenaria*

Birds

Bobwhite quail	*Colinus virginianus*
Chuckar partridge	*Alectoris graeca*
Cowbird	*Melothris ater*
Mallard duck	*Anas platyrhynchos*

Birds (cont.)

Common Name	Scientific Name
Mourning dove	Zenaidura macruora
Pigeon	Columba livia

Insects

Damselfly	Ischnura sp.
Stonefly	Pteronarcys californica

Appendix IV

GLOSSARY

Absorption:	The incorporation of one substance within another.
Acclimatization:	Adjustment of an organism to changes in its environment.
Acute toxicity:	A single dose has more effect than the same quantity administered in several applications.
Adaptation:	An anatomical, behavioural or physiological change in an organism which better enables it to survive in its environment.
Adsorption:	The adhesion of one substance to the surface of another.
Aerobic:	Associated with the presence of free oxygen.
Algae:	A group of simple chlorophyll-containing plants, mostly aquatic; although most are microscopic, some forms reach extremely large sizes.
Algicide:	A chemical specifically toxic to algae, often used to control algal blooms.
Anadromous fish:	Fish spending most of their lives in the sea, but ascending freshwater rivers to spawn.
Anaerobic:	Occurring in the absence of free oxygen.
Anoxic:	Depleted of free oxygen; anaerobic.
Antagonistic:	Causing reduction of toxicity of another chemical.
Application factor:	A factor applied to the results of a short-term toxicity test to estimate the safe concentration of a substance or mixture of substances in water.
Assimilation:	The transformation of absorbed nutrients into body substances or the process whereby a body of water purifies itself of organic pollution.

Bacteria:	Microscopic organisms lacking chlorophyll and a definite nucleus, living either as single cells or in filaments.
Benthic region:	The bottom of a body of water; the organisms inhabiting the benthic region are referred to as the benthos; they contribute to the character of the bottom.
Benthos:	The organisms living at the bottom of a sea, lake, river or estuary; they may be attached to the bottom, creep along its surface of burrow into it.
Bioaccumulation:	Uptake and retention of environmental substances by an organism from its environment.
Bioassay:	The laboratory determination of the effects of substances or conditions upon specific living organisms.
Biochemical oxygen demand (BOD):	A measure of the amount of oxygen consumed (mg/l) in the biological processes that break down organic matter in water.
Biodegradable:	Able to be decomposed readily by the action of micro-organisms.
Biological monitoring:	The use of living organisms to test the quality of waters.
Biomass:	The weight of all life or of a given population in a specified area.
Biota:	All the living organisms within a certain area.
Bloom:	A readily visible proliferation of phytoplankton, macrophytes or zooplankton in a body of water.
Blue-green algae:	A group with a blue pigment in addition to the green chlorophyll, often associated with blooms and capable of fixing atmospheric nitrogen.
Body burden:	The total amount of a substance present in the body tissues and fluids of an organism.
Brackish water:	Water with salinity less than that of sea water, but greater than that of fresh water.

Buffer strip:	A zone left untreated; usually at the outer margin of the treated area or adjacent to streams.
Carcinogenic:	Producing cancer.
Carnivore:	An animal which eats other animals.
Catadromous fish:	Fish spending most of their lives in fresh water but spawning at sea.
Chemical oxygen demand (COD)	A measure of the amount of oxygen required (mg/l) to chemically oxidize organic matter in the water.
Chlorophyll:	The green pigments of plants which enable them to use the energy of the sun for photosynthesis.
Chronic toxicity:	A single dose has less effect than the same quantity of toxin applied in several smaller doses.
Coarse fish:	Those fish species not desirable as game fish or food fish.
Cold-blooded animals (poikilothermic animals):	Animals lacking a temperature-regulating mechanism, whose temperature fluctuates with that of their environment.
Colloid:	Very small particles (< 2 microns) which tend to remain suspended and dispersed in liquids.
Consumer:	An organism that consumes either other organisms or organic food material.
Cultural eutrophication:	The acceleration by man of the build-up of nutrients in a body of water; this build-up is part of the natural aging process of lakes, etc., but normally occurs quite slowly.
Cumulative:	Brought about or increased in strength by successive additions.
Daphnia:	Water fleas, minute crustaceans which are valuable fish food.
Detritus feeder:	An animal which feeds upon organic detritus.
Dispersant:	A chemical agent, often a detergent, used to break up concentrations of organic material, e.g. oil spills.

Diversity:	The abundance of numbers of species in a specified location.
Ecology:	The relationships of living things to one another and to their environment.
Ecosystem:	A biological community together with the physical and chemical resources of its location.
Effluent:	Waste water, treated or not, discharged into bodies of water.
Emergent aquatic plants:	Plants rooted at the bottom and projecting above the surface of the water.
Enrichment:	The addition of nutrients to a body of water.
Enteric:	Of or originating in the intestinal tract..
Environment:	The sum of all external influences and conditions.
Epiphytic:	Living on the surface of a plant.
Estuary:	A partially enclosed coastal water body where tidal effects are evident and fresh water mixes with sea water.
Eutrophication:	The natural process of aging of lakes and other water bodies, involving nutrient enrichment and eventually leading to the drying up of the body of water (see cultural eutrophication).
Eutrophic waters:	Waters with a rich supply of nutrients typically characterized by blooms of aquatic plants, low water transparency and a low dissolved oxygen content.
Evapotranspiration:	The combined loss of water from a given area during a specified period of time by evaporation from soil or water surface and transpiration from plants.
Exchange capacity:	The total ionic charge of the adsorption complex active in the adsorption of ions.
Fauna:	Animal life.
Floating aquatic plants:	Macrophytes that wholly or in part float on the surface of the water (water lilies, etc.).

Flocculation:	The treatment process by which suspended colloidal or very fine particles are assembled into larger masses or floccules which eventually settle out of suspension.
Flora:	Plant life.
Food chain:	The transfer of food energy from producers through a series of consumers, usually four or five.
Food web:	A series of interconnecting food chains.
Fry:	Newly hatched fish; sac fry until the yolk sac is absorbed; then advanced fry until 2.5 cm in length.
Fungi:	Plants without chlorophyll which rely on organic nutrients.
Game fish:	Fish sought by sports fisherman.
Green algae:	Algae with pigments similar in color to those of the higher plants.
Habitat:	The place where the organism lives.
Herbicide:	A pesticide chemical used to destroy or control the growth of undesirable plants.
Herbivore:	An animal which feeds on plants.
Heterotrophic organism:	See consumer.
Homothermic animals:	See warm-blooded animals.
Hypolimnion:	That region of a stratified lake extending below the thermocline to the bottom of the lake.
Impoundment:	A body of water confined by a dam, dike, floodgate or other barrier.
Insecticide:	Pesticide substance intended to repel or destroy insects.
Intertidal:	The area along the shore of a sea or estuary betweed the levels of high and low water.
Invertebrates:	Animals without backbones.

LC_{50}:	That concentration of a particular substance in a suitable diluent (experimental water) at which just 50% of the test organisms are able to survive for a specified period of exposure.
LD_{50}:	That dose of a substance lethal to just 50% of the test organisms within a specified time period.
Lethal:	Involving a stimulus or effect causing death directly.
Life cycle:	The series of stages an organism passes through during its life time.
Limnology:	The study of the physical, chemical and biological aspects of inland waters.
Littoral zone:	The region along the shore of a body of water.
Load:	The amount of a nutrient or other substance discharged into a body of water.
Lysimeter:	A device to measure the quantity or rate of water movement through or from a block of soil, usually undisturbed and in situ or to collect such percolated water for quality analysis.
Macro-organisms:	Organisms visible to the unaided eye.
Macrophytes:	Plants visible to the unaided eye.
Make-up water:	Water added to boiler, cooling tower, or other system to maintain the volume of water.
Mass median diameter:	Droplet diameter such that half the weight of a spray deposit is in larger and smaller droplets.
Median lethal concentration:	See LC_{50}.
Median lethal dose:	See LD_{50}.
Meromictic lake:	A lake in which complete vertical mixing does not occur.
Mesotrophic:	Having a moderate nutrient load resulting in moderate productivity.
Metabolites:	Products of metabolic processes.

Methaemoglobin:	A non-functional haemoglobin produced by the reaction of oxyhaemoglobin with nitrite.
Micronutrient:	Chemical element necessary in only small amounts for growth and development, a trace element.
Micro-organisms:	Minute organisms invisible or barely visible to the unaided eye.
Motile:	Capable of spontaneous movement.
Necrosis:	The death of cellular material within the body of an organism.
Nutrients:	Substances essential as raw materials for the growth of organisms.
Oncogenesis:	Development of cancerous tissue.
Open water:	Water areas unprotected by overhanging vegetation.
Organic detritus:	The particulate remains of disintegrated plants and animals.
Overturn:	The physical phenomenon of vertical mixing which occurs in a body of water following the breakdown of stratification.
Parasite:	An organism living on or in the host organism and obtaining nourishment at the expense of its host for all or part of its life cycle.
Parr:	A young fish, usually a salmonid, between the larval stage and the time it begins migration to the sea.
Pathogenic:	Causing or capable of causing disease.
Periphyton:	Aquatic organisms attached or clinging to plants or other surfaces projecting above the bottom of a lake or stream.
Pesticide:	Any substance used to kill organisms; includes herbicides, insecticides, algicides, fungicides and others.
Photosynthesis:	The process by which simple carbohydrates are produced from carbon dioxide and water by living plant cells, with the aid of chlorophyll and in the presence of light.

Phytobenthos:	The plant life of the benthos.
Phytoplankton:	The plant life of the plankton.
Phytotoxic:	Poisonous to plants.
Piscicide:	A substance used to destroy or control fish populations.
Plankton:	Organisms of relatively small size that swim weakly or drift with the water masses.
Poikilothermic animals:	See cold-blooded animals.
Producers:	Organisms which synthesize organic substances from inorganic substances.
Productivity:	The rate of production of organic material.
Pulse contamination:	A short-lived higher concentration of pollutant followed by a rapid decline to a much lower concentration or complete absence thereof.
Recharge:	To add water to the zone of saturation, as in recharge of an aquifer.
Reducers:	Organisms which digest food outside the cell by means of enzymes and then absorb the food into the cell and reduce it to inorganic matter.
Release:	To eliminate competition of undesirable vegetation from established trees.
Reservoir:	A pond, lake, tank or basin, natural or man-made, used for the storage or control of water.
River basin:	The total area drained by a river and its tributaries.
Runoff:	That portion of precipitation or irrigation water which flows across the ground surface, eventually returning to the streams.
Safety factor:	A numerical value applied to short-term data from other organisms in order to approximate the concentration of a substance that will not harm or impair the organism being considered.
Saprophytic:	Living on decayed organic matter.
Secondary treatment:	See biological (secondary) treatment.

Sessile organisms:	Organisms which rest on or are attached to a substance, without a supporting stalk.
Seston:	All the particulate matter suspended in water.
Sorption:	Absorption or adsorption.
Species (singular and plural):	A natural, reproductively isolated, population or group of populations which transmit specific characteristics from parent to off-spring.
Spodic horizon:	A layer in some acid soils in which colloidal oxides of iron and aluminum have accumulated. These oxides can tie up large amounts of phosphorus.
Standing crop:	See biomass.
Stratification:	Separation into layers.
Sublethal:	Involving a stimulus below the level that causes death.
Submerged aquatic plant:	A macrophyte which is continuously submerged beneath the surface of the water.
Substrate:	The underlying material on which an organism moves or to which it is attached.
Subtidal:	Below the level of low water in a sea or estuary.
Succession:	The sequence of communities which replace one another in a given area until a relatively stable community becomes established.
Surfactant:	A surface active agent; a component of detergents.
Symbiosis:	Two organisms of different species living together, with benefit to one or both and harm to neither.
Synergism:	The combination of the effects of separate substances such that the total effect is greater than the sum of the individual effects.
Teratogen:	A substance causing birth defects.
Thermocline:	The layer in a body of water where the temperature difference is greatest per unit depth.

Threshold dose:	The minimum dose of a substance necessary to produce a measurable effect in the test organism.
Tolerance:	Capacity to endure an environmental factor.
Toxicity:	Potency of a toxic or poisonous substance or combination of substances.
Trophic level:	Position in the food chain (i.e. producer, consumer, etc.).
Turbidity:	Cloudiness of the water.
Vertebrates:	Animals with backbones.
Warm and cold-water fish:	Groups of fish distinguished by the adaptation of the eggs to development in warm or cold water.
Warm-blooded animals (homothermic animals):	Animals with a temperature regulating mechanism capable of maintaining a nearly constant body temperature, independent of environmental temperature fluctuation.

Adapted from "A compilation of Australian water quality criteria" Technical Paper No. 7. Australian Water Resources Council, 1974. Barry T. Hart.